Section 7

Effects of the Action

This section evaluates the potential impacts of routine City of Seattle projects on ESA-listed species and their critical habitat within the Seattle action areas. Routine project activities, which are described in Section 3 and categorized as 14 construction methods, can have short-term construction-related impacts and long-term impacts as an affected habitat stabilizes to new project features.

7.1 Effects of the Action on the Species

Under the Seattle Biological Evaluation, the conservation measures described in Section 4 will be incorporated into proposed projects that use the routine construction methods. While these conservation measures will reduce potential impacts to listed species and/or their critical habitat, the measures will vary with each project.

The following effects of the action are described in this section. The construction methods that result in these effects are shown on Table 7-1:

- Vegetation removal: riparian area
- Sediment
- Stream isolation and fish removal
- Pile removal
- Pile driving
- Overwater structures
- Vactoring and excavation
- Shoreline hardening, bank stabilization, and habitat enhancement and restoration activity
- Culvert replacement
- Boating activity
- Pesticides.

Table 7-1
Effects of action and corresponding methods and conservations measures for the Seattle Biological Evaluation

Effects of action	Method	Conservation measures #
Vegetation removal: riparian area	2. Clearing, grubbing, grading and placement of temporary fill	1, 7, 9, 12-14, 19, 20, 23, 72, 74
Sediment	2. Clearing, grubbing, grading and placement of temporary fill	1, 7, 9, 12-14, 19, 20, 23, 72, 74
	3. Temporary dewater of upland construction site	1, 30
	4. Work area isolation and fish removal in streams, large waterbodies and for pipe bypass	31, 32
	5. Pipe and culvert installation, replacement and maintenance	1-4, 12-18, 30, 75
	6. Vactoring, jetting, excavating accumulated sediments	1-4, 15-18, 21, 25-29, 54, 55, 57-59, 62
	7. Bank stabilization	1-4, 9, 15-18, 27-29, 43-52, 54-62, 64, 66-71
	8. Habitat addition and maintenance	1-7, 9-22, 25, 26, 28-30, 54-62, 65, 66, 72-74
	9. Trench safety/support systems	1-4, 15-21, 25, 26, 30, 54, 55
	10. Beach nourishment/ substrate addition	1, 4, 15, 16, 27, 29, 54-56, 59, 63
	11. Boat launch improvement, repair, and maintenance	1-3, 15, 16, 18, 28, 29, 54-56, 60, 66
	12. In-water/overwater structure repair and replacement	
	12**A**. Piling	**A**. 1, 34, 43-53, 59, 62
	12**B**. Anchor and chain systems	**B**. 1, 15, 16, 29, 41, 42, 59, 62
	12**C**. Superstructure, decking etc.	**C**. 1, 3, 4, 7, 12, 15, 16, 18, 19, 25-29, 33, 35, 37, 38, 43, 44, 46, 52, 54-56, 59-62
	12**D**. Floats and Gangways	**D.** 1-4, 6, 7, 9, 12-19, 25, 26-29, 33, 35-37, 60-62
	12 **E**. Floating log boom	**E.** 1, 39, 41, 42, 59, 62
	12**F**. Buoys	**F.** 1, 41, 42, 59-62
	12**G**. Fixed breakwaters	**G.** 1-4, 12-20, 27,28, 40, 54-56, 59-62, 64, 70, 71
	13. Site restoration	1, 4, 11, 12, 15, 16, 18, 19, 24, 54, 55, 62
Stream isolation and fish removal	4. Work area isolation and fish removal in streams, large waterbodies and for pipe	31, 32

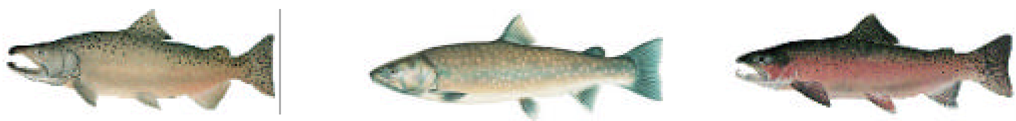
	bypass	

Table 7-1
Effects of action and corresponding methods and conservations measures for the Seattle Biological Evaluation

Effects of action	Method	Conservation measures #
Pile removal	12. In-water/overwater structure repair and replacement	See above under *Sediment*
Pile driving	7. Bank stabilization 12. In-water/overwater structure repair and replacement	See above under *Sediment*
Overwater structures	11. Boat launch improvement, repair and maintenance 12. In-water/overwater structure repair and replacement	See above under *Sediment*
Vactoring and excavation	5. Pipe and culvert installation, replacement and maintenance 6. Vactoring, jetting, excavating accumulated sediment	See above under *Sediment*
Shoreline hardening, bank stabilization, habitat enhancement and restoration activity	8. Habitat addition and maintenance 9. Trench safety / support systems 10. Beach nourishment and substrate addition 11. Boat launch improvement, repair and maintenance 12. In-water/overwater structure repair and replacement	See above under *Sediment*
Culvert replacement	4. Work area isolation and fish removal in streams, large waterbodies and for pipe bypass 5. Pipe and culvert installation, replacement and maintenance 6. Vactoring, jetting, excavating accumulated sediments	See above under *Sediment*
Boating activity	11. Boat launch improvement, repair and maintenance 12. In-water/overwater structure repair and replacement	See above under *Sediment*
Pesticides	13. Site restoration	See above under *Sediment*
	14. Landscape and planting	1-7, 9-16, 18-20, 22, 25, 26, 54, 60-62, 64, 69-73, 75

Some listed species such as the humpback whale and marbled murrelet, rarely occur within the Seattle action areas.[1] Therefore, the effects of the proposed actions on these species are not addressed in this document. Other species such as the killer whale and Steller sea lion are found within the marine waters of the City of Seattle, although in low numbers. Certain project activities such as pile driving may affect killer whales and Steller sea lions if they are in the area. The effects of the action are described for these 2 species were applicable (i.e., increased sound pressure levels).

7.1.1 Puget Sound Chinook Salmon, Bull Trout, and Steelhead



7.1.1.1 Effects of Vegetation Removal: Riparian Area

Removal of trees and vegetation within the riparian zone has several impacts or alterations to watershed conditions and capacity. The primary pathways for negative impacts are through altering stream temperature patterns, hydrologic and sediment regimes, and reducing the structural features that maintain channel complexity.

Removing trees and vegetation from the riparian zone can lead to numerous impacts to listed fish, their habitat, and prey species by the following (Spence et al. 1996):

- Reducing stream shading and channel stability
- Elevating fine sediments in spawning gravels and filling of substrate interstices
- Reducing pool habitats
- Altering nutrient balance and physical character of the stream
- Reducing cover and overall stream habitat complexity
- Restricting natural movements of juveniles and adults.

Stream Temperature

Water temperatures significantly affect the distribution, health, and survival of fish, specifically salmonids in streams. Because these fish are ectothermic (cold-blooded), their survival depends on external water temperatures. They will experience adverse health effects when exposed to temperatures outside their optimal range (USEPA 2003).

[1] These species may be found in the Elliot Bay, North Seattle/Puget Sound, and South Seattle/Puget Sound action areas. However, because these species have been rarely documented within the marine waters of the City, potential impact to these species from proposed activities will be negligible.

Adverse temperatures can affect growth, behavior, disease resistance, competition, and mortality (Sullivan et al. 2000).

Removal of trees and vegetation near streams affects stream temperatures primarily 2 ways: 1) reducing streamside canopy levels and 2) increasing exposure of upland soil surfaces to solar radiation.

The potential for riparian vegetation to mediate stream temperatures is greatest for small-to-intermediate size streams and diminishes as streams increase in size, lower in the floodplain (Spence et al. 1996). Generally, small and intermediate streams represent most of the total aggregate stream length within a watershed (Chamberlin et al. 1991). Given these relationships, maintaining adequate canopy conditions on small- and medium-sized streams (including intermittent ones) is necessary to avoid altering natural temperature regimes.

Groundwater entering streams (especially small streams) is an important determinant of stream temperatures (Spence et al. 1996) and provides localized thermal refugia in larger stream systems. Where groundwater flows originate above the neutral zone (52 to 59 feet [16-18 m] below the surface in general) groundwater temperatures vary seasonally, influenced by air temperature patterns (Spence et al. 1996).

Project activities such as site preparation, clearing and grubbing, bank stabilization, etc. may slightly impact stream temperatures through loss of shade resulting from removal of trees and vegetation within riparian buffers. These impacts to stream temperature will be minimal because of the small amount of riparian buffers lost for such activities and minimization measures implemented to avoid or reduce loss of riparian vegetation during the project. However, the amount of riparian buffers is limited in the Seattle action areas, and any loss of vegetation will increase stream temperatures within the watershed.

Hydrologic Alterations

Hydrologic and sediment regimes are altered by vegetation removal, site disturbance, and soil compaction associated with construction activities (USDA and USDI 1998, Keppeler 1998). The nature and magnitude of these changes are moderated by local climatic, geologic, and topographic characteristics as well as revegetation patterns (Spence et al. 1996).

Removal of vegetation typically reduces water loss to evapotranspiration, resulting in increased water yield from the watershed and enhanced base flows (Spence et al. 1996, Keppeler 1998). Increases in peak flows following vegetation removal have been reported. They are likely the result from combined effects of vegetation removal and more rapid routing of water from uplands to the stream channel.

Channel Complexity

Tree and vegetation removal can alter processes that create and maintain riparian and aquatic habitats, often reducing habitat complexity and aquatic species diversity (Elmore and Beschta 1987, USDA et al. 1993, USDA and USDI 1998). Changes in habitat features associated with reductions in habitat complexity include decreases in large woody debris, pool quality, channel stability, substrate quality, groundwater inflows, and suitable habitat serving as corridors between habitat patches (MBTSG 1998, Spence et al. 1996).

Hardwoods have replaced conifers in many urban riparian areas that humans have altered or managed. Woody debris produced by deciduous vegetation tends to be smaller, more mobile, and shorter-lived than that derived from conifers and does not function as well in retaining sediment (Spence et al. 1996). Reduced supply of large woody debris decreases channel stability and leads to a loss of instream cover and pool habitat and decreased retention of sediments, including gravels used by salmonids for spawning, and simplifies channel hydraulics.

In many City projects, while riparian vegetation is removed for access to the stream, stream restoration activities including placement of boulders, large woody debris or other bioengineered techniques are used to increase channel stability and complexity. These techniques, while engineered, provide increased salmonid habitat much faster than what would occur naturally. In addition, site restoration (Method 13) and landscaping and planting (Method 14) activities are used to repair or replant disturbed areas. These activities establish vegetation along the stream faster and reduce the long-term impact of vegetation removal.

Conservation Measures

In delineating the work area, the City will identify and protect environmentally sensitive areas including riparian corridors. Construction areas will be defined on project plans and flagging will be used to mark off areas at the project site. Construction impacts, including clearing and grubbing, will be confined to the minimum area necessary to complete the project. Vegetation will be retained to the maximum extent possible. In addition, when temporary fill is needed to access or work platforms, timber mats, pallets, hog fuel (wood waste), or other biodegradable material will be used to minimize total removal of vegetation. See Table 7-1 for corresponding construction methods and conservation measures for the effects of vegetation removal in the riparian corridor.

7.1.1.2 Effects of Sediment

The following activities may result in sediment inputs in the Seattle action areas:

1. Excavation above the wetted perimeter
2. Restoring streamflow on the reconstructed streambed
3. Disturbance of the bank and riparian area by construction and restoration activities.
4. Discharge of water back into stream following dewatering of upland site or during sediment removal or excavation projects.
5. Post-project channel adjustment or stabilization.

Sedimentation Effects to the Aquatic Environment

The introduction of sediment can have multiple effects on channel conditions and processes resulting in effects on listed fish and prey species survival, the food web, and water quality conditions, such as water temperature and dissolved oxygen (Rhodes et al. 1994). Fine sediments can influence incubation survival and emergence success (Weaver and White 1985 in MBTSG 1998). Emergence success depends on the quantity and size of the fine sediment within spawning gravels. Table 7-2 summarizes the maximum percentage of fines of different sizes that corresponds to 50% emergence for different salmonids. In general, the smaller the fines (< 0.83 mm) the smaller percentage of fines needed to reduce emergence by 50%.

Sediment can modify stream morphology and function through the following (Bash et al. 2001):

- Degradation of spawning and rearing habitat
- Simplification and damage to habitat structure and complexity
- Loss of habitat
- Decreased connectivity between habitats.

Biological implications of this habitat damage can include the following (Newcombe and Jensen 1996):

- Underutilization of stream habitat
- Abandonment of traditional spawning habitat
- Displacement of fish from their habitat
- Avoidance of habitat.
- Egg/fry mortality.

As sediment enters a stream it is transported downstream under normal fluvial processes and deposited in areas of low shear stress (MacDonald and Ritland 1989). These areas are usually behind obstructions, near banks (shallow water) or within interstitial spaces. This episodic filling of successive storage compartments continues in a cascading fashion downstream until the flow drops below the threshold required for sediment to move or all pools have reached their storage capacity (MacDonald and Ritland 1989). As sediment loads increase, the stream compensates by geomorphologic changes such as increased slope, increased channel width, decreased depths, and decreased flows (Castro and Reckendorf 1995). These processes increase erosion and sediment deposition.

In addition, social behavior patterns may be altered by suspended sediment (Berg and Northcote 1985). High concentrations of suspended sediment can also affect survival, growth, and behavior of stream biota that are forage for salmonids (Harvey and Lisle 1998). Suspended sediment may alter the food supply by decreasing abundance and availability of aquatic insects. However, the precise thresholds are difficult to characterize for fine sediment in suspension or in deposits that result in harmful effects to benthic invertebrates (Chapman and McLeod 1987).

Substrate embeddedness is an indicator of the overall habitat condition and is evaluated at the stream-reach scale. Within a reach of a given stream, rearing habitat is considered to be 'functioning' at various levels as follows (NFMS 1996, USFWS 1998):

- Appropriately when reach embeddedness is less than 20%
- At risk when reach embeddedness is 20 to 30%
- At unacceptable risk when reach embeddedness is more than 30%.

Table 7-2
Maximum percentage of fines corresponding to 50% emergence for salmonids

Species	Maximum percentage of grains finer than:				Reference
	0.83 mm	**2.0 mm**	**3.35 mm**	**6.35 mm**	
Brook trout		10			Hausle and Coble 1976
Chinook salmon				15, 26 40 30, 35	Bjorn 1969 Tappel and Bjornn 1983 McCuddin 1977
Chum salmon				27	Koski 1975; 1981
Coho salmon	7.5, 17 21 11		 30 36		Cederholm and Salo 1979 Phillips et al. 1975 Koski 1966
Cutthroat trout				20	Irving and Bjornn 1984
Rainbow trout	 12			30 40	Irving and Bjornn 1984 NCASI 1984
Steelhead				25 39 27	Bjornn 1969 Tappel and Bjornn 1977 McCuddin 1977
			25		Phillip et al. 1975
Source: Kondolf 2000					

The addition of fine sediment to streams during the summer decreased abundance of juvenile Chinook salmon in almost direct proportion to the amount of pool volume lost to fine sediment (Bjornn et al. 1977, Bash et al. 2001). Similarly, the density of rearing Chinook salmon was inversely related to the abundance of fine sediment, illustrating the importance of winter habitat containing low sediment loads (Bjornn et al. 1977). As fine sediments fill the interstitial spaces between the cobble substrate, juvenile Chinook salmon were forced to leave preferred habitat and to utilize cover that may be more susceptible to ice scouring, predation, and decreased food availability (Hillman et al. 1987). Deposition of sediment on gravel substrates also may lower winter carrying capacity for bull trout (Shepard et al. 1984) and the abundance of aquatic invertebrates, an important food source for young salmonids.

Eggs and alevins are generally more susceptible to stress caused by suspended solids than are adults. Egg survival is dependent on a continuous supply of well-oxygenated water through the streambed gravels (Cederholm and Reid 1987). Accelerated sedimentation can reduce the flow of water and, therefore, oxygen to eggs and alevins. That in turn can decrease egg survival, decrease fry emergence rates (Cederholm and Reid 1987, Chapman 1988, Bash et al. 2001), delay development of alevins (Everest et al. 1987), and reduce growth and cause premature hatching and emergence (Birtwell 1999). Fry delayed in their timing of emergence are less able to compete for environmental resources than other fish that have undergone normal development and emergence (intra- or interspecific competition) (Everest et al. 1987).

Whether eggs/alevins are smothered or fry emergence is impeded is largely determined by sediment particle sizes of the spawning habitat (Bjornn and Reiser 1991). Sediment particle size determines the pore openings in the redd gravel and with small pore openings, more suspended sediments are deposited and water flow is reduced compared to large pore openings.

Several studies have documented that fine sediment can reduce the reproductive success of salmonids. Natural egg-to-fry survival of coho salmon, sockeye and kokanee has been measured at 23%, 23%, and 12%, respectively (Slaney et al. 1977). Substrates containing 20% fines can reduce emergence success by 30 to 40% (MacDonald et al. 1991). A decrease of 30% in mean egg-to-fry survival can be expected to reduce salmonid fry production to low levels (Slaney et al. 1977).

Due to in-water timing restrictions (work windows) for instream construction, sediment will be generated at a time with least impact to fish life-history stage. However, spawning habitat and active redds may be impaired by unavoidable post-construction sediment entering the river from areas disturbed by construction. If this occurs, sediment deposited on redds could result in egg and alevin mortality, particularly where existing levels of fine sediment (less than 6.4 mm [0.25 in]) in the streambed (embeddedness) are high. Fish movement may also be temporarily obstructed by increased suspended sediment due to construction and post-construction sedimentation caused by precipitation.

Conservation Measures

Temporary erosion and sediment control measures are required on all projects to minimize sediment input into the stream and other sensitive areas. These measures include, in part, covering excavated and stockpiled material, placing sediment barriers (silt fences, wattles, etc.) around disturbed sites, and placing erosion control measures over disturbed areas. Proper rewatering of streams after construction will also minimize sediment effects downstream of construction sites. A stepwise rewatering of the site will prevent sudden increase in downstream turbidity. See Table 7-1 for corresponding construction methods and conservation measures for the effects of sediment.

7.1.1.3 Effects of Stream Isolation and Fish Removal

Proposed routine project activity may result in impacts to fish from specific construction elements such as the following:

- Capture and transport of fish
- Block nets
- Seines, dip nets and minnow traps
- Electroshocking
- Stream dewatering.

Capture and Transport of Fish

To reduce lethal impacts on listed fish species from dewatering the stream, the City of Seattle proposes to capture and relocate all fish from project construction sites before construction begins. The City of Seattle proposes using seines and dip nets, block nets, and electroshocking. Although this effort will reduce the overall impact to endangered, threatened, and proposed species, fish may in some cases experience immediate or delayed injury or death from the use of nets and/or electroshocking techniques. Most of the injuries and death will be due to block nets and electroshocking. Mortality associated with handling stress, seine, and dip nets is unlikely. The City of Seattle proposes releasing all captured fish and aquatic organisms as close to the point of capture as possible. Other considerations for releasing fish will be based on their life-history stage and number of fish captured. Juvenile fish will be released downstream of the site to aid migration out of the system. Adult fish will be released upstream to aid migration to spawning or resting locations. All fish will be released in the best available habitat to reduce or decrease predation and aid recovery.

Block Nets

Before dewatering a stream section, block nets will be placed up- and downstream from the work area to prevent fish entering the stream segment that will be dewatered. The use of block nets poses a mortality risk to fish, even when monitored daily.

In 2000, the U.S. Fish and Wildlife Service studied bull trout sampling efficiency in Washington, capturing 811 bull trout (2,364 salmonids total) with block nets (J. Polos, USFWS, pers. comm. 2001). Total fish mortality was 92 (4% of the total captured). Bull trout accounted for 63% of all mortalities (n=58) and 7% (58 of 811) of all bull trout captured died on the block nets due to impingement. All bull trout mortalities were either fry (n=47) or juveniles (n=11).

To potentially reduce the level of mortality risk, the City of Seattle proposes monitoring block net use in a slightly different manner than that of the U.S. Fish and Wildlife Service study. The U.S. Fish and Wildlife Services' collection methods in the 2000 study resulted in stream flows continually passing through the block nets throughout the night with crews checking nets one time during the night.

Under the proposed action of the Seattle Biological Evaluation—which are mostly maintenance projects—the City of Seattle will install block nets, capture and relocate all fish, divert streamflow around the project area, then remove the block nets all in the same day. On rare occasions, block nets may remain in the stream overnight when the fish capture and diversion activities require additional time to complete.

However, block nets normally will be installed and removed the same day (during daylight hours). Personnel will be available during the day to remove fish promptly, thus avoiding long-term/lethal impacts of fish impingement on block nets. In addition, stream dewatering will occur during authorized in-water work timing windows which will minimize potential impingement to listed and proposed fish species. Therefore, the impingement of fish will be rare and result in significantly less mortality when compared with methods used for the U.S. Fish and Wildlife Service 2000 study.

Seines, Dip Nets, and Minnow Traps

Seines and dip nets will initially be used to capture and remove any fish trapped between the block nets in the portion of the stream to be dewatered. The use of seines and dip nets is expected to capture about 70% of the fish within the section of stream to be dewatered. However, this is highly site specific, as sites with large woody debris and undercut banks are difficult to seine or use dip nets.

Minnow traps involve using wire-mesh traps placed in key instream fry habitat overnight before dewatering. Captured fish are then removed and relocated either upstream or downstream based on the fish life history stage. Fry will be transported in large buckets (minimum 5 gallon [19 L]) filled with stream water. The fish and water temperature will be monitored to ensure the health of the fish until they are released. Given the low impact of these capture and relocation techniques, fish are not expected to be injured by the method.

Electroshocking

The capture and handling of Puget Sound Chinook salmon, bull trout, and steelhead through electroshocking is a short-duration activity, occurring intermittently over a single day. However, electroshocking may result in a high risk of mortality and injury including spinal hemorrhages, internal hemorrhages, fractured vertebra, spinal misalignment, and separated spinal columns (Hollender and Carline 1994, Dalbey et al. 1996, Thompson et al. 1997).

Electroshocking has been found to have a high rate of injury to fish. Factors that influence fish injuries include environmental conditions (conductivity of water, depth of water, or substrate), electrical hardware, and the electrical current (Sharber and Carothers 1988). Voltage, pulse shape, and frequency are electrical factors causing fish injuries (Sharber and Carothers 1988, McMichael 1993, Dalbey et al 1996). Table 7-3 summarizes studies on the effects of electroshocking on fish.

Table 7-3
Summary of effects of DC electroshocking on fish

Fish species	Percent with spinal injury	Percent with hemorrhage injury	Conservation measures #
Rainbow trout	22*, 45**	34*, 45**	Thompson et al. 1997
Brown trout	32*, 36**	24*, 35*	Thompson et al. 1997
Rainbow trout			Dalbey et al. 1996
Smooth DC	12		
Half-pulse DC	40		
Full-pulse DC	54		
Brook trout	17	16	Holinder and Carline 1994
Rainbow trout			McMichael 1993
300 v, smooth DC	4	4	
300 v, 30 Hz	22	35	
300 v, 90 Hz	35	53	
400 v, smooth DC	14	17	
Rainbow trout			Sharber et al. 1994
Anode type			
sphere	43		
cable	65		
ring	43		
Pulse frequency			
15 pulses/sec	3		
30 pulses/sec	24		
60 pulses/sec	43		
512 pulses/sec	62		
Burst of waves	8		
Rainbow trout			Sharber and Carothers 1988
Pulse shape			
exponential	44		
square	44		
quarter size	67		

*Shore-based pulsed-DC equipment
**Boat electroshocking pulsed-DC equipment

Spinal injuries in fish ranged from 3% to 67% which depended on the voltage, pulse shape, and frequency used during electroshocking. Smooth DC or low frequency DC (< 30 Hz) electroshocking results in less injury to fish. Hollender and Carline (1994) found most spinal injuries were either rating class 2 (40%) or 3 (40%) (Table 7-4). They also found the spinal injuries involved on average 7 vertebrae, and were usually located in the region of the spinal column between the dorsal and anal fins.

While electroshocking has significant effects on injury to fish, the degree of spinal injury does not affect long-term survival (Dalbey et al. 1996). There is an influence on growth. Rainbow trout with moderate and severe spinal injury (classes 2 and 3) grew little in length and weight after 335 days (Dalbey *et al.* 1996). Thompson *et al.* (1997) speculated that fish in better condition may be more likely to be injured because of more powerful muscle contractions.

Dalbey et al. (1996), Thompson et al. (1997), and Hollender and Carline (1994) all found longer fish had a higher probability of being injured. Incidence and severity of injury were positively correlated with fish length: 40% of rainbow trout longer than 8 inches (20 cm) sustained injury compared with 27% in smaller fish (Dalbey et al. 1996). The injury rate was the following (Hollender and Carline 1994):

- Lowest (12%) for brook trout smaller than 5 inches (12.7 cm)
- Intermediate (26%) for the 5- to 7-inch length (12.7-17.8 cm) group
- Highest (43%) for the 7-inch-and-longer length (17.8 cm) group.

Table 7-4
Rating system to identify and rate severity of electroshocking injuries

Rating class	Internal hemorrhage	Spinal damage
0	None apparent	None apparent
1	Mild hemorrhage with 1 or more wounds in the muscle, separate from the spine	Compression (distortion) of vertebrae only
2	Moderate hemorrhage with 1 or more small wounds on the spine (<= width of 2 vertebrae)	Misalignment of vertebrae, including compression
3	Severe hemorrhage with 1 or more large wounds on the spine (> width of 2 vertebrae)	Fracture of 1 or more vertebrae or complete separation of 2 or more vertebrae

Source: Thompson et al. 1997

Very few of the fish collected by Thompson et al. (1997) exhibited external signs of injury although a higher percentage of rainbow and brown trout were injured by electroshocking than would have been suspected from external examination. Dalbey et al. (1996) found that rainbow trout X-rayed soon after capture, exhibited no detectable signs of spinal injury, but later showed calcification indicative of old injuries when X-rayed again after 335 days in a pond. Hollender and Carline (1994) found hemorrhages and spinal compressions in the smallest fish were small and difficult to see and might have been overlooked. Therefore, their reported injury rate (average of 22%) may be a conservative estimate. In addition, most studies have focused on injuries exhibited by adults, but stress from electroshocking can be the main problem for juveniles (P. Bisson, U.S. Forest Service, S. Parmenter, California Department of Fish and Game, pers. comm. in Nielson 1998).

The City of Seattle uses Smith Root LR-24 backpack electroshockers that are capable of adjusting voltage (50 to 990 v), pulse shape (smooth, pulsed, or burst), and frequency

(1 to 120 Hz) (Smith-Root website at www.smith-root.com). The Smith Root LR-24 electroshockers also has an automatic initial set-up system. This system automatically sets the electroshocker to the current stream conditions. This set-up gives a good starting point to minimize impacts to fish. When proper electroshocking techniques are used, potential fish injury is minimized. Proper electroshocking techniques are identified in the NMFS Electroshocking Guidelines. In addition, all stream dewatering and fish handling will occur during approved in-water work windows, which minimize the potential to injure proposed or listed fish. No large (subadults or adults) proposed or listed fish species should be in any of the action areas during the in-water work window, especially in City streams. Juvenile proposed or listed species may be present in some streams during the in-water work window, but the City has never captured a proposed or listed species during stream dewatering (G. Lockwood, City of Seattle, pers. comm. 2006). Rainbow trout are captured in some streams within the action areas (Thornton and Longfellow creeks). These fish have been identified as rainbow trout and not steelhead.

Stream Dewatering

During stream dewatering—including when sandbags are used to focus stream flows—there is a potential that a few fish may avoid being captured and relocated, and thus may die because they remain undetected in stream margins under vegetation or gravels. A gradual dewatering approach, as proposed, should enhance the efficacy of fish removal and thus reduce, but not eliminate, this risk. An estimated 95% of the fish will be removed before total dewatering of the stream. In addition, due to the proposed timing of the activities, the risk to listed fish species should be minimized because of the reduced likelihood of migratory and/or spawning fish being in the stream reach during construction.

Conservation Measures

Method 4 of this Seattle Biological Evaluation is the method to isolate the in-water construction site. This method in the past has been a conservation measure to reduce, minimize, or avoid potential effects to fish. It has now become a routine practice. In addition, this method in conjunction with work timing windows has greatly reduced construction-related impacts to fish. See Table 7-1 for corresponding construction methods and conservation measures for the effects of stream isolation and fish removal.

7.1.1.4 Effects of Pile Removal

Projects proposing to remove creosote-treated timber piles by either full extraction or breaking off or cutting the piles at or below the mudline will result in temporary suspension and a long-term increase in creosote-contaminated sediments within the project area. There are 2 potential pathways for increased long-term contamination that could result from this practice:

1. The first pathway is waterborne. Waterborne (surface water and water column) sediment contamination can appear when piles are pulled out or cut. The creosote

on the pile's surface has been buried in an anoxic zone and is essentially fresh creosote and highly volatile when re-exposed. Freshly-cut piles generally act in a wicking fashion, pulling the fresh creosote from within the pile and from sediments in the anoxic zone toward the freshly-cut surface. This fresh creosote can be suspended in the water column as well as increase contamination of the sediment.

2. The second pathway consists of droplets of fresh creosote released from the piles into surrounding sediments as piles are being pulled. Because these droplets are heavier than water, they sink to the bottom and very likely are undetectable in the water column. Puget Sound Chinook salmon, bull trout, and steelhead could be directly exposed to contaminants suspended in the water column or indirectly exposed through the food chain.

Creosote contains numerous constituents known to be toxic to aquatic organisms (Eisler 1987, Germain et al. 1993, Brooks 1995, Van Brummelen et al. 1998, Brooks 2000, Johnson et al. 2002). Creosote is composed primarily of PAHs (about 65-85%), with smaller percentages of phenolic compounds (10%), and nitrogen-, sulfur-, or oxygenated heterocyclics (Brooks 1995, EPRI 1995). PAHs are introduced into the environment through industrial discharges, creosote from treated woods, municipal runoff, and atmospheric emissions from incineration and automobile emissions. PAHs are also introduced into the marine ecosystems through accidental spills of fuel oil and other petroleum products.

The general mode of effect associated with acute exposure to PAHs is non-polar narcosis (Van Brummelen et al. 1998). Other major effects include biochemical activation/adduct formation (carcinogenesis), phototoxicity (acute and chronic exposure), and disturbance of hormone regulation. The role of PAHs in endocrine disruption is not well documented. Immunotoxicity as a mode of PAH toxicity has been investigated (Varanasi et al. 1993, Karrow et al. 1999). PAHs have induced tumors in laboratory animals exposed by inhalation and ingestion (Germain et al. 1993). The presence of hepatic (liver) tumors in English sole (*Parophrys vetulus*), a benthic marine fish, has been linked to PAH contamination in sediments collected from industrialized areas around Puget Sound (Krahn et al. 1986, Meyers et al. 1990, Stein et al. 1990, Johnson et al. 2002).

In addition to liver disease, fish (specifically English sole) residing in Puget Sound areas with elevated levels of PAHs have been documented to suffer from various types of reproductive impairment. This impairment includes inhibited ovarian development, inhibited spawning ability and reduced egg viability (Johnson et al. 2002). Moreover, exposure to PAHs in the water column caused flatfish larvae to become disoriented and to exhibit signs of narcosis, while higher concentrations of PAHs were associated with increased mortality (Johnson et al. 2002).

Schirmer et al. (1999) evaluated the cytotoxicity and photocytotoxicity of intact and photomodified creosote to rainbow trout gill cell cultures. The study found that high creosote doses were necessary to elicit a cytotoxic response in rainbow trout gill cell cultures. The toxic potency of creosote to rainbow trout gill cell cultures was strongly influenced by UV radiation. UV irradiation of either the creosote or the creosote-exposed cell cultures consistently increased the toxicity of creosote to fish gill cells in culture and, at least in the case of the photocytotoxicity of creosote, was attributable to PAHs.

Karrow et al. (1999) reported depression of biological indicators for immune function in rainbow trout that had been exposed to liquid creosote in microcosms. Immune function was evaluated in juvenile Chinook salmon collected from contaminated waterways around Puget Sound and compared with hatchery fish caught upriver (Varanasi et al. 1993). Compared with reference fish from hatcheries located upstream on the Green and Puyallup rivers, fish from the Duwamish Waterway and the Commencement Bay/Puyallup River estuary had elevated concentrations of PCBs and aromatic hydrocarbons in body tissues and stomach contents. The fish from the estuaries exhibited immunosuppression in comparison with the hatchery fish, as indicated by tests of humerol and cellular-mediated immunity.

Studies have shown that high concentrations of toxic chemicals in sediments are adversely affecting Puget Sound biota via detritus-based food webs (NOAA 2000, Johnson et al. 2002). PAHs, introduced into the marine system through sources such as petroleum product spills or creosote treated wood, tend to adsorb to sediments. When sediment is undisturbed, only a portion of parent PAH compounds are readily bioavailable to marine organisms. However, resident benthic organisms may be exposed to PAHs through their diet, through exposure to contaminated water in the benthic boundary layer, and through direct contact with the sediment. PAHs may bioaccumulate in aquatic invertebrates within these benthic communities (Varanasi et al. 1989, Meador et al. 1995). Therefore, benthic invertebrate prey are a significant source of PAH exposure for marine fish. Vertebrate organisms are able to quickly metabolize some of the lighter PAH compounds and readily excrete a percent of the hydrophobic parent compound along with the polar water-soluble metabolites (James et al. 1991, McElroy et al. 1991), which can be passed on to consuming marine fish. While PAHs do not bioaccumulate in vertebrates, some heavier, more carcinogenic PAH compounds and metabolites may persist and are known to cause sub-lethal effects to fish exposed in laboratory studies (NTP 1999) and field studies (Moore and Myers 1994, Myers et al. 1998a, 1998b, O'Neill et al. 1998).

Acute and chronic toxicity have been evaluated in laboratory experiments for a variety of aquatic organisms (i.e., mysids (*Mysidopsis bahia*), oysters (*Crassostrea virginica*), pink shrimp (*Penaeus duorarum*), Mosquito fish (*Gambusia affinis*), Dungeness crab larvae (*Cancer magiste*r), coho salmon, and rainbow trout (Brooks 1995, BOR 2000). Application of these laboratory results to real-world exposures is difficult because the release of PAHs into natural waterbodies from treated wood (i.e., environmental exposure) differs significantly from the methods used to introduce pure compounds into essentially sterile laboratory conditions (Poston 2001). The high number of variables contributes to a high level of uncertainty in understanding the risk for exposure and the potential effects.

Environmental exposure to creosote and PAHs depends on the age of the treated wood, methods used to treat the product, a host of environmental parameters, and dilution by the receiving waterbody. Chinook salmon, bull trout, steelhead, and prey fish species could become exposed to creosote and its associated contaminants (i.e., PAHs) when old pilings are removed. They will be directly exposed to these constituents while they are suspended in the water column. Given the chemical composition and characteristics of creosote (i.e., in general this chemical and associated compounds are hydrophobic and will adsorb to particulate matter in the water column and later settle out into bottom sediment [Johnson et al. 2002]) the waterborne creosote concentrations should be negligible within a week of re-suspension (J. Davis, USFWS, pers. comm. 2004). Levels of creosote and PAH exposure would probably not be high enough to cause direct cytotoxicity or tumor induction but may cause immune suppression in Chinook salmon, bull trout, steelhead, and prey fish species, resulting in increased disease susceptibility.

Puget Sound Chinook salmon, bull trout, and steelhead could be indirectly exposed to contamination through the foodchain. Creosote and associated chemicals remaining in sediments at the site and wherever they settle out after suspension are likely to persist for many years given their resistance to biological breakdown. Creosote and its chemical constituents have a half-life of about 3 years in biological components (e.g., water, soils). The length of persistence will depend on the concentration of chemicals added to the environment during the removal of the piles, which is currently unknown. As creosote and associated chemicals are known to bioaccumulate in invertebrates, Puget Sound Chinook salmon, bull trout, and steelhead are expected to be exposed to creosote/PAH compounds through the foodchain. Over the long-term, with the treated timber piles removed and, therefore, the source of creosote removed or capped by the sediment falling into the hole left by the extracted pile, the following happens:

- The concentration of creosote in the sediment will decrease
- Water quality will improve
- The pathway of exposure for fish through contamination of prey will be reduced.

We anticipate that direct exposure, in the water column, and indirect exposure, through the foodchain, will affect individuals. These effects could result in reduced reproductive success (e.g., inhibited ovarian development, inhibited spawning ability, and reduced egg viability) and reduced survival (e.g., impacts resulting in cytotoxicity, tumors, immune suppression, etc.). However, we expect that a significant impairment of breeding, feeding or sheltering activities or the normal behaviors associated with these activities will be difficult to discern at the individual level. At this point, impairment may only be detectable at the population level (i.e., declines in population). In the long term, removal of creosote piles is expected to improve water quality for Puget Sound Chinook salmon, bull trout, steelhead, and their prey species by decreasing concentrations in the water column and chronic contamination of benthic invertebrates.

Conservation Measures

If treated piles are fully extracted or if they are cut below the mudline, the City will cap the holes or piles with appropriate material such as clean substrate (sand and/or gravel) or pile caps. This ensures that chemicals from the existing piles do not leach into the adjacent sediments or the water column. See Table 7-1 for corresponding construction methods and conservation measures for the effects of pile removal.

7.1.1.5 Effects of Pile Driving

Projects involving the impact installation or proofing of steel piles will result in effects to Puget Sound Chinook salmon, bull trout, steelhead, and prey species through underwater sound pressure levels. During pile installation, either an impact or a vibratory hammer pile driver will be used. In some circumstances both pile drivers may be used. An impact hammer is a large piston-like device that is usually attached to a crane. A vertical support holds the pile in place and a heavy rod moves up and down, striking the top surface of the pile. A vibratory hammer has a set of jaws that clamp onto the top of the pile. The pile is held by the jaws while the hammer vibrates the pile. The vibrations liquefy the surrounding sediments and the combined weight of the hammer and pile cause it to sink to the desired depth. Piles that are installed with a vibratory hammer often must be 'proofed.' Proofing involves striking the pile with an impact hammer to determine the load-bearing capacity of the pile and usually involves multiple strikes. Juvenile and adult Puget Sound Chinook salmon and steelhead and sub-adult and adult bull trout may be affected from in-water impact and vibratory pile driving.

High underwater sound pressure levels are known to have negative physiological and neurological effects on a wide variety of vertebrate species including fish and birds (Yelverton et al. 1973, Yelverton and Richmond 1981, Steevens et al. 1999, Fothergill et al. 2001, Cudahy and Ellison 2002, USDOD 2002). High underwater sound pressure levels are known to injure and/or kill fish by causing barotraumas (pathologies associated with high sound levels including hemorrhage and rupture of internal organs), as well as causing temporary stunning and alterations in behavior (Turnpenny et al. 1994, Turnpenny and Nedwell 1994, Popper 2003, Hastings and Popper 2005). Risk of injury appears related to the effect of rapid pressure changes, especially on gas-filled spaces in the bodies of exposed organisms (Turnpenny et al. 1994).

High underwater sound pressure levels can also cause several behavioral responses that have not been well studied. Broadly, the effects of elevated underwater sound pressure levels on organisms range from death to no effect. Over this continuum of effect, there is no easily identifiable point at which behavioral responses transition to physical effects. A number of technical acoustic descriptors are used throughout this section (Table 7-5).

From a point source in a uniform medium (such as water), sound spreads outward following common laws of Transmission Loss physics (i.e., spherical or cylindrical spreading laws). Transmission Loss physics implies that intensity and pressure vary inversely with the square of the distance from the source. With spherical spreading, sound pressure levels diminish by about 6 dB when the distance is doubled. For cylindrical spreading, sound pressure levels diminish by about 3 dB with every doubling of distance. Sound transmission in shallow water is highly variable and site specific. Refraction can result in either reduced or enhanced sound transmission in shallow water (Richardson et al. 1995). Therefore, a practicable spreading loss (Davidson 2004) provides a more accurate analysis on reduction of sound pressure levels through water. The practicable spreading model uses a 4.5 dB reduction with every doubling of distance from the source.

Table 7-5
Acoustic concepts and terminology

Term	Definition
Sound	Vibrations in air, water, etc., that stimulate the auditory nerves and produce the sensation of hearing. The perception of a sound depends on 2 physical characteristics: 1) amplitude and 2) frequency. Both can be measured.
Amplitude	Measure of the acoustic energy of sound vibrations. Sound amplitude is measured on a logarithmic scale in units called decibels.
Frequency	Rate of oscillation or vibration of sound measured in cycles per second, or hertz (Hz). Ultrasonic frequencies are those that are too high to be heard by humans (greater than 20,000 Hz). Infrasonic sounds are too low to be heard by humans (less than 20 Hz).
Decibel (dB)	Numerical expression of the relative loudness of a sound. The reference scale for underwater sound is 1 micro-pascal (µPa) and is expressed as "dB re: 1µa". A pascal (Pa) is the pressure resulting from a force of 1 newton exerted over an area of 1.2 square yards (1 m^2).
Sound pressure levels (SPL)	Sound pressure that is expressed in dB. In this document, underwater sound pressure levels are referred to in units of dB. Peak pressure (peak): The highest level or amplitude or greatest absolute sound pressure level during the time of observation. Sound pressure levels expressed as peak are used in discussing injury or mortality to fish. Root mean square (rms) is the root square of energy divided by the duration. Sound pressure levels expressed as rms are commonly used in discussing behavioral effects. Behavioral effects—which often result from auditory cues and effects on hearing—may be better expressed through averaged units rather than by peak pressures.
Impulse	Measure of the total energy content of the pressure wave. Positive impulse is the integral of pressure over time measured from the arrival of the leading edge of the pressure wave until the pressure becomes negative.
Transmission loss (TL)	Loss of sound energy, expressed in dB, as sound passes through a medium like water. Several factors are involved: the spreading of the sound over a wider area (spreading loss), losses to friction (absorption), scattering and reflections from objects in the sound's path, and interference with 1 or more reflections of the sound off of surfaces (for underwater sound, the surfaces are substrate and air-water interface).

Impact Pile Driving: Underwater Noise Effects Resulting in Injury or Mortality

Impact pile driving of steel piles can produce intense sound pressure levels that are known to injure and/or kill fish (Stotz and Colby 2001, Stadler 2002, Fordjour 2003, Abbott et al. 2005, Hastings and Popper 2005). Injuries of this type are referred to as barotraumas, and include hemorrhage and rupture of internal organs, swim bladder rupture, hemorrhaged eyes, and temporary stunning (Yelverton et al. 1973, Yelverton and Richmond 1981, Turnpenny and Nedwell 1994, Hastings and Popper 2005). Death from barotrauma can be instantaneous, occurring within minutes after exposure, or several days later (Abbott et al. 2002).

The potential for injury to fish or any other aquatic organism from pile driving depends on the type and intensity of the sounds produced. These are greatly influenced by a variety of factors, including the type of hammer, the type of substrate, and the depth of the water. Firmer substrates require more energy for pile driving, and produce more intense sound pressures. Biologically, key variables that factor into the degree to which an animal is affected include size, anatomical variation and location in the water column (Office of Naval Research 1998). Any gas-filled structure within an animal is particularly susceptible to the effects of underwater sound (Office of Naval Research 1998). Examples of gas-filled structures in vertebrate species are swimbladders, bowel, sinuses, lungs, etc. As a sound travels from a fluid medium into these gas-filled structures, there is a dramatic drop in pressure that can cause rupture of the hollow organs (Office of Naval Research 1998). Sound energy from an underwater source readily enters the bodies of animals because the acoustic impedance of aquatic animal tissue nearly matches that of water (Hastings 2002). This has been demonstrated in fish with swimbladders (such as salmonids). As a sound pressure wave passes through a fish, the swimbladder is rapidly compressed due to the high pressure and then rapidly expanded by the underpressure component of the wave. At the high sound pressure levels associated with pile driving, the swimbladder may repeatedly expand and contract, hammering the internal organs that cannot move away since they are bound by the vertebral column above and the abdominal muscles and skin that hold the internal organs in place below the swimbladder (Gaspin 1975). This pneumatic pounding can also rupture capillaries in the internal organs, as observed in fish with blood in the abdominal cavity, and maceration of kidney tissues after exposure to pile driving (Abbott et al. 2002, Stadler 2002). Hastings (Ohio State University, pers. comm. 2003) also noted that the differential vibration of the various tissues can cause tearing when the pressure wave passes through the fish.

Physical injury to aquatic organisms may not result in immediate mortality. If an animal is injured, death may occur several hours or days later, or injuries may be sublethal. Necropsy results from Sacramento blackfish (*Othodon microlepidotus*) exposed to high sound pressure levels showed fish with extensive internal bleeding and a ruptured heart chamber were still capable of swimming for several hours before death (Abbott et al.

2002). Sublethal injuries can interfere with the ability to carry out essential life functions such as feeding and predator avoidance (Popper 2003).

Another mechanism of injury and death resulting from high sound pressure levels is ‘rectified diffusion,’ or the formation and growth of bubbles in tissue. Rectified diffusion can cause inflammation and cellular damage because of increased stress and strain (Vlahakis and Hubmayr 2000, Stroetz et al. 2001) and blockage or rupture of capillaries, arteries, and veins (Crum and Mao 1996). Crum and Mao (1996) analyzed bubble growth underwater by rectified diffusion caused by sound signals at low frequencies (less than 5,000 Hz), long pulse widths, and atmospheric pressure. Their analysis indicated that sound pressure levels exceeding 210 dB_{peak} could cause bubble growth within a few seconds, but bubble growth was not expected at sound pressures below 190 dB_{peak}.

Yelverton et al. (1973) and Yelverton and Richmond (1981) exposed many fish species, various birds and terrestrial mammals to underwater explosions. Common to all species exposed to underwater blasts were injuries to air and gas-filled organs, as well as eardrums. These studies identified injury thresholds in relation to size of the charge, range at which the charge was let, and mass of the animal exposed. For fish, Yelverton et al. (1973) and Yelverton and Richmond (1981) found that the greater the mass (weight of the fish), the greater impulse level needed to cause an injury. Conversely, a smaller mass would sustain injury from a smaller impulse.

At Bremerton, Washington, approximately 100 surfperch (*Cymatogaster aggregata*, *Brachyistius frenatus* and *Embiotoca lateralis*) were killed during impact driving of 30-inch (76 cm) diameter steel pilings (Stadler 2002). The size of these fish ranged from 2.7 to 6.9 inches (70-175 mm) FL. Dissections revealed that the swimbladders of the smallest of the fish (3.1 inch [80mm] FL) were completely destroyed, while those of the largest individual (6.7 inches [170 mm] FL) were nearly intact. Damage to the swimbladder of *C. aggregata* was more severe than to similar-sized *B. frenatus*. These results are suggestive of size and species-specific differences and are consistent with those of Yelverton et al. (1975), who found size and/or species differences in injury from underwater explosions. Due to their size, adult salmonids can likely tolerate higher sound pressure levels (Hubbs and Rechnitzer 1952, Yelverton et al. 1975), and their injury rates are expected to be less than those of juvenile fish.

The most noticeable and documented effects of pile driving have been fish kills. However, it is important to note that not all fish killed by pile driving float to the surface. At the Port of Vancouver, British Columbia, divers found that a large number of dead fish, including salmonids, had sunk to the bottom (WSDOT 2003). Teleki and Chamberlain (1978) found that up to 43% of the fish killed by underwater explosions sank to the bottom. With few exceptions, fish kills are reported only when dead and injured fish are observed at the surface. Thus, the frequency and magnitude of such kills are likely underestimated.

Small fish that are subjected to high sound pressure levels may also be more vulnerable to predation, and the predators themselves may be drawn into the potentially harmful field of sound by following injured prey. The California Department of Transportation reported that the stomach of a striped bass killed by pile driving contained several freshly consumed juvenile herring. It appears this striped bass was feeding heavily on killed, injured, or stunned herring that swam into the zone of lethal sound pressure.

Implications and Extent of Underwater Sound Resulting in Injury or Mortality

Hastings (2002) stated that little to no physical damage to aquatic animals can be expected from peak sound pressures below 190 dB. However, much uncertainty exists as to the level of adverse effects to fish exposed to sound between 180 and 190 dB_{peak} due to species-specific variables. Turnpenny et al. (1994) exposed brown trout (*Salmo trutta*) to pure tone bursts of sound pressure levels greater than 170 dB_{peak} at 500 Hz or less for 90 seconds. This resulted in a mortality rate of up to 57% after 24 hours. Similar studies with whiting (*Merlangius merlangus*) resulted in 50% mortality occurring at sound pressure level greater than 176 dB_{peak} (95 Hz) (Turnpenny et al. 1994). They suggest that the threshold for eliciting various effects on fish for continuous sounds was lower than for pulsed sounds such as seismic airgun blasts. Impact pile driving is expected to generate underwater sound that is more similar to seismic airguns than to pure tone bursts. Based on this information the 170 dB_{rms} threshold for injury to brown trout identified by Turnpenny et al. (1994) is likely to be lower than the injury threshold level anticipated for impact pile driving.

Based on the preceding information, the potential for mortality and injury is expected to occur at sound pressure levels greater than or equal to 180 dB_{peak}. The 180 dB_{peak} threshold is conservative because most of the studies described evaluated transmitted signals of longer duration than is anticipated to result from pile driving.

To estimate the geographic area in which mortality and injury would be expected, the distance at which Transmission Loss attenuates the pressures to below the thresholds must be estimated. Calculating Transmission Loss is extremely complicated, and is likely to be site-specific. A practical spreading model, as described by Davidson (2004) [Transmission Loss = 15*Log(R)] can be used to estimate the distances at which mortality, injury and behavioral disruption are expected. This model assumes that sound pressure levels decrease at a rate of 4.5 dB per doubling distance.

Impact Pile Driving: Underwater Noise Impacts Resulting in Behavioral Disruption

This section addresses only those effects that could result in behavioral disruption. It summarizes existing information and its application to effects on Puget Sound Chinook salmon, bull trout, and steelhead.

Most of the sound energy of impact hammers is concentrated at frequencies between 100 and 800 Hz. Salmonids are thought to have optimal hearing at frequencies of 150 Hz (Hawkins and Johnstone 1978). However, they are known to detect sounds at frequencies of up to 600 Hz (Mueller et al. 1998), and as low as 10 Hz (Knudsen et al. 1992).

Therefore, impact pile installation produces sounds within the range of salmonid hearing and at frequencies that have been demonstrated to trigger a behavioral response.

Sounds that reach their peaks rapidly tend to elicit stronger alarm reactions from fish than sounds with equal peak intensities but longer rise times (Schwartz and Greer 1984). Popper (2003) suggests that behavioral responses to loud sounds may include the fish temporarily swimming away from the sound source, thereby decreasing the potential exposure to the sound, or the animal 'freezing' and staying in place, thereby leaving the fish open to possible injury.

Alternatively, responses to sound could affect behavior more extensively and result in fish leaving a feeding ground (Engås et al. 1996) or an area in which it would normally reproduce or in some other way affect long-term behavior and subsequent survival and reproduction. Popper (2003) suggests that the effect of these avoidance responses may range from insignificant, to a more permanent long-term effect if feeding or reproduction is impeded.

Feist et al. (1992) found that impact pile driving of concrete piles affected juvenile pink and chum salmon distribution, school size, and schooling behavior. In general, on days when pile driving was not occurring, the fish behaved in a polarized schooling pattern, moving in a definite pattern. When pile driving was occurring, the fish exhibited an active milling schooling behavior (moving in an eddying mass). Fish appeared to change distributions about the site, orienting and moving towards an acoustically-isolated cove side of the site on pile-driving days more than on non-pile driving days.

Knudsen et al. (1992) studied spontaneous awareness reactions (consisting of reduced heart beat frequency and opercular movements) and avoidance responses to sound in juvenile Atlantic salmon. This study evaluated the responses of these fish to frequencies ranging from 5 to 150 Hz. With increasing frequencies, the difference between the threshold for spontaneous awareness reaction and the estimated hearing threshold also increased. At 5, 60, and 150 Hz, the signal had to exceed the hearing thresholds by 25, 43, and 73 dB, respectively, to elicit the reactions.

Sound has also been used to try and control the migration of salmonids (Carlson 1994). Burner and Moore (1962) were not able to elicit any response from rainbow trout and brown trout to sounds between 67 Hz and 70 kHz at sound levels of up to 182 dB. Moore and Newman (1956) did not observe a response from young salmonids when they used sounds from 50 to 200 kHz at high sound levels. Other investigators (cited by Van Derwalker 1967) attempted to use sound to guide juvenile Chinook salmon and striped bass (*Morone saxatilis*) through a sound barrier (frequency and sound level were not reported). Although animals in a laboratory showed some response to the sounds, there was no apparent response in a river.

Turnpenny et al. (1994) studied brown trout, bass, sole, and whiting and found that behavioral sensitivity is lowest in flatfish, which have no swimbladder, and also in salmonids (brown trout), in which a swimbladder is present, but somewhat remote from the inner ear. This study found that with brown trout an avoidance reaction occurred above 150 dB_{rms} and that other reactions (e.g., a momentary startle), were only noted at 170 to 175 dB_{rms}. Additionally, the study references Hastings' 'safe limit' recommendation of 150 dB_{rms} and concludes based on his study that the safe limit provides a reasonable margin below the lowest levels where fish injury was observed. Observations by Feist et al. (1992) suggest that sound levels in this range may also disrupt normal migratory behavior of juvenile salmon.

Fewtrell (2003) held fish in cages in marine waters and exposed them to seismic airgun impulses. The study detected significant increases in behavioral responses when sound pressure levels exceeded 158 to 163 dB_{rms}. Responses included alarm responses, faster swimming speeds, and tighter groups and movement toward the lower portion of the cage. It is difficult to discern the significance of these behavioral responses. The study also evaluated physiological stress response by measuring plasma cortisol and glucose levels and found no statistically significant changes. Conversely, Santulli et al. (1999) found evidence of increased stress hormones after exposing caged European bass to seismic survey noise.

Most information on the behavioral effects of underwater sound is from studies that use pure tone sound. Sounds generated by pile driving, however, are made up of multiple frequencies/tones, making comparisons difficult. With regard to the effect of different frequencies within the 0 to 3,000 Hz band, Turnpenny et al. (1994) recommended that all frequencies be treated as equally potentially harmful to fish. They also suggested that such an approach would capture both the range of swimbladder resonant frequencies likely to be found in fish of different sizes and species (determining susceptibility to injury), and the hearing sensitivities of different species, (determining avoidance behavior). Current information regarding the salmonid hearing range suggests that salmon can detect sounds up to 600 Hz (Mueller et al. 1998), but become less sensitive to hearing above 380 Hz (Hawkins and Johnstone 1978). Therefore the range (0-3,000 Hz) suggested by Turnpenny et al. (1994) for salmonids is likely overestimated.

Based on the above, sound pressure levels in excess of 150 dB_{rms} are expected to cause temporary behavioral changes, such as elicitation of a startle response or avoidance of an area. They are not expected to cause direct permanent injury.

Vibratory Pile Driving: Review and Assessment of Existing Information and Data

Adverse effects in the form of physical injury or mortality, or behavioral disruption to Puget Sound Chinook salmon, bull trout, and steelhead from vibratory pile driving is not expected. This assumption is based on the significant differences, discussed here, in the underwater sounds produced by vibratory driving of piles when compared with those from impact driving of piles.

The sounds from vibratory hammers differ from those of impact hammers not only in intensity, but in frequency and impulse energy (total energy content of the pressure wave). Most of the sound energy of impact hammers is concentrated between 100 and 800 Hz—the frequencies thought most harmful to aquatic animals—while the sound energy from the vibratory hammer is concentrated around 20 to 30 Hz. Additionally, during the strike from an impact hammer, sound pressure rises much more rapidly than during the use of a vibratory hammer (Carlson et al. 2001, Nedwell and Edwards 2002).

Just as these 2 sounds differ, so do the observed behavioral responses of fish to them. Most of the energy in the sounds produced by vibratory hammers is at the frequency of vibration, around 20 to 30 Hz, near the range of infrasound (less than 20 Hz). Fish have been shown to avoid infrasound (Knudsen et al. 1997), but not sounds at 150 Hz (Enger et al. 1993, Dolat 1997, Knudsen et al. 1997, Sand et al. 2000), and habituation to the sound does not occur, even after repeated exposure (Dolat 1997, Knudsen et al. 1997). However, depending on the location of the vibratory installation, sound pressure levels may not exceed ambient sound levels. Vibratory installation of steel piles in a river in California resulted in sound pressure levels that were not measurable above the background noise created by the current (Reyff 2006).

Since the sounds from vibratory hammers are near the frequency of infrasound, and are of long duration, they may elicit an avoidance response (Knudsen et al. 1997, Carlson et al. 2001). In laboratory tanks, Knudsen et al. (1997) observed that infrasound (10 Hz) evoked flight and avoidance responses in juvenile Pacific salmon (*O. tshawytscha* and *O. mykiss*). Additionally, fish avoided the area (within [3.3 feet [1 m]) close to the sound source when it was continuously running. However, this behavioral response differs from that expected during impact hammering. Fish may react to the first few strikes of an impact hammer with a 'startle' response. After these initial strikes, the startle response wanes and fish may remain within the field of a potentially harmful sound (Dolat 1997, NOAA Fisheries 2001).

Vibratory hammers produce peak pressures that, in general for all pile sizes, are about 17 dB lower than those generated by impact hammers (Nedwell and Edwards 2002). Although this may result in sound pressure levels above those expected to cause physical injury, the sounds generated by vibratory hammers differ in intensity, frequency and impulse energy. These differences may account for the fact that no fish kills have been associated with use of vibratory hammers. The lack of evidence does not mean that vibratory installation is completely harmless, but due to difference in the waveform, vibratory installation appears less harmful than impact installation.

Impact Installation of Concrete and Wood Piles

The effects of impact installation of concrete and wood piles are not well documented or understood. Hydroacoustic monitoring has been conducted for installation of concrete (Abbott et al. 2005, Anderson and Reyff 2006) and wood (Carlson et al. 2001) piles during impact pile driving. Monitoring of 24-inch (61 cm) concrete piles resulted in sound pressure level of 185 to 189 dB_{peak} and 18-inch-square (116 cm^2) concrete piles had sound pressure levels of 175 to 180 dB_{peak} (Anderson and Reyff 2006).

Hydroacoustic monitoring of 12-inch (30.5 cm) diameter wood piles resulted in sound pressure levels of 195 dB_{peak}.

Limited data indicate that impact installation of concrete and wood piling results in a slower accumulation of energy and generally lower sound pressure levels compared to installation of steel piling (Rodkin and Donavan 2004, Abbott et al. 2005, Anderson and Reyff 2006). Data from studies of blasting indicate that the shape of the sound pressure wave is an important factor in determining whether an organism may be physically injured by the pressure wave (Hastings and Popper 2005). Pressure waveforms where the initial peaks are steep and rise quickly are considered more likely to cause potential injury compared to pressure waveforms with slower rise times on the initial peak (Yelverton et al 1975, Hastings 2002, Wardle et al. 2001). Therefore, one might assume that installation of concrete and wood piles may be less injurious than installation of steel piles. It is important to note that it is difficult to compare data from blasting to pile driving for many reasons. One notable difference between the pile driving and blasting is that blasting presents a single impulse while pile driving is repetitive.

During impact installation of piles, the pile is struck repeatedly, and multiple strikes may occur over a 1-second period. Monitoring of wood pile installation on a project revealed that the pile was struck at a rate of 2 blows per second (Carlson et al. 2001). Monitoring data from impact driving of steel piles have indicated that there is an additive effect from repetitively striking a pile, and that the sound energy increases throughout the pile-driving event (Abbott et al. 2002).

It is possible that impact installation of concrete and wood piles could result in behavioral responses potentially affecting Puget Sound Chinook, bull trout, and steelhead migratory and foraging patterns. Salmonids are thought to have optimal hearing at frequencies of 150 Hz (Hawkins and Johnstone 1978). However, they are known to detect sounds at frequencies of up to 600 Hz (Mueller et al. 1998), and as low as 10 Hz (Knudsen et al. 1997). In laboratory tests, exposure to sounds at 10Hz evoked flight and avoidance responses in Pacific salmonids (Knudsen et al. 1997).

Sound generated by impact installation of wood piles includes a very low frequency component that may be due to lateral movement of the pile after it is hit with the hammer (Carlson et al. 2001). Although most energy of the impulse for wood piles in this study was contained at frequencies around 200 Hz and higher (Carlson et al. 2001), the low frequency component is within the range shown to trigger a behavioral response (Knudsen et al. 1997). These behavioral responses could disrupt normal feeding and migratory behavior. There may be a long-term effect if feeding is impeded (Popper 2003). Engas et al. (1996) found that exposure to elevated sound pressure levels from air guns resulted in declines in fish catch rates that lasted for several days after air gun use. It is unknown whether the fish left the area or if they were injured or killed, but it is important to note the effects may last longer than the pile driving. Conversely, fish may ‘freeze’ and stay in one place, increasing the potential for physical effects such as hearing loss and injury. Normal ‘fright’ response of many fish is to freeze (Popper 2003).

Elevated sound pressure levels from pile driving may prevent fish from hearing biologically relevant sounds. As a result, a fish may be more vulnerable to predation, or conversely, decrease their ability to find prey (Popper 2003). Because fish gain important environmental information from sound, anything that hampers the ability to detect biologically relevant signals could have a deleterious effect on the survival of fish and the health of fish populations (Popper 2003).

Factors to consider in evaluating the potential behavioral effects of concrete and wood pile installation include the duration of the work, diurnal timing, and location (e.g., near a forage fish base). Significant physical effects to listed fish may not occur from installation of concrete and wood piles because the sound pressure wave generated from impact pile driving of concrete and wooden piles is different from steel piles, and no fish kills have been documented during their installation.

Reducing Underwater Sound Pressure Levels

A sound attenuation system, such as a pile 'cap,' bubble curtain, or combination of both, may be used to reduce sound pressure levels and to lengthen rise times.

Pile caps are typically wood, or nylon discs placed between the pile hammer and the top of the pile. Caps have long been used by pile driving contractors to protect the pile from damage. Effectiveness varies depending on the material used. In 2006, Washington State Parks compared effectiveness between 4 pile cap materials: wood, nylon, Combest and Micarta (Laughlin 2006). Hydroacoustic monitoring during impact installation of 12-inch (30.5 cm) steel piles with the 4 cap types showed that wood caps reduced sound pressure levels more than caps made from the other materials (average reduction with the wood cap was 24 dB). Use of a wood cap also lengthened rise times. For example, on 1 pile, the rise time was 1.8 msec without the wood cap and was 37.7 msec with the wood cap. Other materials did not lengthen rise times to this degree. (Laughlin 2006)

Use of a bubble curtain can be an effective method for reducing sound pressure levels from pile driving. The degree of effectiveness depends on the design as well as the site conditions. Spacing of the bubble manifolds, air pressure, tidal currents, and water depth are all factors influencing effectiveness. Improper installation and operation can also decrease bubble curtain effectiveness in reducing sound pressure levels (Visconty, Anchor Environmental, pers. comm. 2004, Pommerenck 2006).

Studies on the effectiveness of bubble curtains on reducing sound pressure waves have found varied results. Longmuir and Lively (2001) report reductions of more than 17 dB resulting from use of a carefully designed bubble curtain. Use of a U-shaped bubble curtain around a steel pile wave barrier also yielded reductions up to 17 dB (Laughlin 2006). Vagle (2003) evaluated the underwater effects of pile driving at 4 locations in Canada and obtained reductions of between 18 and 30 dB when using a properly designed bubble curtain. Reyff (2003) reviewed previous reports, and also conducted an additional study on the use of bubble curtains and their reduction of sound pressure waves. In previous studies, Reyff (2003) found that bubble curtains resulted in the following:

1. 0 to 2 dB attenuation in sound pressure levels (due to strong currents)
2. 5 to 10 dB reduction in peak dB
3. 3 to 5 dB reduction in rms.

Vagle (2003) studied the underwater effects of pile driving at 4 locations in Canada. This study reported reductions of between 18 and 30 dB when using a properly designed bubble curtain.

At Friday Harbor, San Juan Islands, Washington, the Washington State Ferries monitored steel pile driving with and without a bubble curtain (Visconty, Anchor Environmental, pers. comm. 2004). Initially, the bubble curtain was improperly installed and no sound attenuation was observed. The bubble curtain was not placed firmly on the bottom; therefore, unattenuated sound escaped under the bubble curtain. After the bubble curtain was modified by adding weight and a canvas skirt to conform to the bottom contour of Puget Sound, the sound was attenuated by up to 12 dB, with an average of 9 dB reduction.

Reyff et al. (2002) evaluated the effectiveness of an isolated pile using a bubble curtain system. The isolated pile was 12.5 feet (3.8 m) in diameter with the interior coated with 1-inch (2.54-cm) closed cell foam. In this type of bubble curtain system, the isolated pile surrounds the actual driven pile, and contains the bubble flow. The isolated pile and bubble curtain system provided a dramatic reduction in both peak pressures and rms levels. Peak pressures were reduced by 23 to 24 dB and rms levels were reduced by 22 to 28 dB. Most of the reduction in sound energy occurred at frequencies above 100 Hz.

Bubble curtains may also minimize injury to fish by changing the shape of the impulse wave. A bubble curtain and a fabric barrier system were both used during a pile installation demonstration project at the San Francisco – Oakland Bay Bridge (Caltrans 2001). The bubble curtain did not attenuate peak sound pressure levels, but changed the shape of the impulse wave, resulting in a more gradual accumulation of energy at the start of pile driving. The overall effect of this on fish is unknown, because fish were still killed and injured with the use of the bubble curtain, although in smaller numbers than without a bubble curtain. The fabric curtain system was found to effectively reduce dB_{rms} values, but no specific numbers in dB reduction were given (Caltrans 2001). The fabric barrier is estimated to reduce sound pressure levels by 10 to 5 dBs [Figure 4-8 in (Caltrans 2001)].

Impact installation of large (7.9 feet [2.4 m] diameter) piles with an isolation casing combined with an air bubble curtain resulted in significant sound pressure attenuation on a project in California. During impact pile driving in the San Joaquin River an attenuation system consisting of an isolation casing with a bubble curtain on the inside achieved much less attenuation (between 6-9 dB) (Pommerenck 2006). However, this project had problems correctly implementing the system. During impact installation of steel piles in an embayment on the Columbia River a bubble curtain built using the Longmuir and

Lively (2001) design achieved a maximum reduction of 17 dB, although the results were variable (Laughlin 2006). A test of bubble curtain effectiveness in Friday Harbor, Washington, found improvements were seen after the original design was modified onsite to improve contact with the substrate. After modification, the bubble curtain was achieving a 12 dB reduction, which equates to an 85% reduction in peak overpressure (Laughlin 2005). Use of a bubble curtain while installing 24-inch (0.6 m) steel piles at a marina in Washington resulted in reductions of 10 to 15 dB (Houghton and Smith 2005).

Conservation Measures

The following conservation measures to minimize sound pressure level impacts will be implemented when installing piles:

- Plastic, cement, or timber piles should be used instead of steel piles
- Vibratory driver should be used as much as possible depending on the load capacity
- Bubble curtain or other noise attenuation method (wood blocks, nylon blocks, etc.) shall be used during impact installation or proofing of steel piles
- Hydroacoustic monitoring is required during installation of 12-inch or larger piles.

See Table 7-1 for corresponding construction methods and conservation measures for the effects of pile driving.

7.1.1.6 Effects of Overwater Structures

Overwater structures in both marine and freshwater alter important habitat controlling factors (light regime, wave energy, substrates, and water quality) that support salmonid and prey species biological and ecological functions such as predator-prey relationships, behavior, spawning, rearing and refugia (Simenstad et al. 1999, Carrasquero 2001, Nightingale and Simenstad 2001a). The nearshore habitat in marine waters and the edge or littoral habitat in freshwater are the most vulnerable areas altered by humans (Brown 1998, Barwick et al. 2004). Incremental impacts of shoreline development through the construction of docks and piers result in cumulative losses of habitat diversity and complexity (Barwick et al. 2004). Direct effects of shoreline development include physical structure alterations of bottom substrate modifications, removal of coarse woody debris, loss and fragmentation of aquatic vegetation, and simplification of shoreline habitat through bulkhead construction (Kelty and Bliven 2003, Scheuerell and Schindler 2004).

A variety of overwater structures line the marine waters of the Seattle action areas. These structures range from residential boat docks to large industrial and commercial piers like ferry terminals and piers along the Seattle waterfront. The effects of docks and piers in marine waters include behavioral responses to fish migration, alteration of light regimes, hydrology and wave energy attenuation, substrate and sediment transportation and distribution, changes in macroinvertebrate density and diversity, and water quality changes (Simenstad et al. 1999, Nightingale and Simenstad 2001a, Haas et al. 2002),

In freshwater, most of the overhead structures are public and residential boat docks. However, in the Lake Washington Ship Canal, large commercial piers have been constructed to moor large boats (fishing, sightseeing, etc.), houseboats, and smaller pleasure crafts. The effects of these docks and piers are similar to those in marine waters. Overwater structures in freshwater effect predator-prey interactions, riparian and aquatic vegetation loss, alterations of light regimes, changes in migration, wave energy alterations, and water quality effects (Kahler et al. 2000, Carrasquero 2001).

Activities during construction of piers, docks and associated bulkheads result in permanent loss or destruction of aquatic and riparian vegetation and woody debris (Kahler et al. 2000, Haas et al. 2002, Kelty and Bliven 2003). Installation of pilings disturbs the substrate and vegetation. The presence of pilings lessens the chance of vegetation regrowth. Pilings, especially in marine waters, alter currents and sediment deposition, which affects vegetation growth (Kelty and Bliven 2003, Williams et al. 2003a).

Overwater structures result in sharp underwater light contrast that affect plant communities, macroinvertebrates and fish populations. Under-pier light energy loss falls below the threshold amounts needed for photosynthesis affecting macrophyte and phytoplankton primary projection (Simenstad et al. 1999, Kahler et al. 2000, Carrasquero 2001, Nightingale and Simenstad 2001a, Williams et al. 2003a). These photsynthesizers are an important part of the marine nearshore habitat and the estuarine and nearshore food webs that support juvenile salmonids and other fish in the nearshore (Simenstad et al. 1999, Nightingale and Simenstad 2001a). Submerged aquatic vegetation and marsh grasses provide important habitat, filter nutrients and sediments, provide nursery habitat for fish, and stabilize bottom sediments (Kelty and Bliven 2003). Increased shading due to overhead structures reduces plant shoot density, biomass, and growth (Kelty and Bliven 2003). Although the area of vegetation loss associated with any individual dock may be relatively small, cumulative impacts and fragmentation of vegetation beds may be significant along highly developed shorelines (Shafer and Robinson 2001). Dock height, width, construction material, and orientation to the sun are primary factors in determining shade effects to vegetation (Nightingale and Simenstad 2001a, Shafer and Robinson 2001, Kelty and Bliven 2003, Williams et al. 2003a).

Fish migration along the shoreline in marine waters and freshwater shows behavioral responses upon encountering docks and piers (Nightingale and Simenstad 2001a). Migrating salmonid responses to docks and piers include migration delays due to disorientation, school dispersal, and migration directional changes (Nightingale and Simenstad 2001a). Salmon fry have been found to migrate along the edges of the shadows of overhead structures rather than penetrate them (Williams et al. 2003a), although this may be species dependent (Williams et al. 2003a). In marine and freshwater environments, as Puget Sound Chinook salmon increased in size, they move further offshore and did not migrate under overwater structures (Ratte and Salo 1985, Tabor et al. 2006).

For marine waters, studies of potential increases in predation of salmonids have not documented any increase in predation associated with overwater structures in the marine environment (Ratte and Salo 1985, Shreffler and Moursund 1999, Nightingale and Simenstad 2001a). Williams et al. (2003a) studied potential salmon predators at Washington State ferry terminals in Puget Sound, and while predators were slightly more abundant at the terminals as compared to unmodified shores, they found no evidence that predation increased at the terminals. In freshwater, predation has been observed near overhead structures (Carrasquero 2001). Overwater structures provide cover for predators and prey, but predators have the advantage because complex habitat that juvenile salmonids need to avoid predators is missing (Barwick et al. 2004). In Lake Washington and the Ship Canal, salmonid predators such as smallmouth and largemouth bass can be found directly under piers (Tabor et al. 2004c, 2006).

In an experimental study of juvenile Chinook salmon, Kemp et al. (2005) found that juvenile Chinook salmon strongly avoided overhead cover. The fish responded to visual cues related to either the presence of an overhead structure or the area of darkness it created. Similarly, Tabor et al. (2006) watched schools of juvenile Chinook salmon as they migrated along the shores of a Lake Washington. As the fish approached a pier, they altered their migration by heading out to deeper water where they either went under or swam around the pier, or on a few occasions, fish appeared to turn around and head in the direction from which they came. These changes in migration patterns may lead to increased energetic demands to the juveniles or increased risk of predation (Kemp et al. 2005).

In freshwater, overhead structure can increase the rate of predation on juvenile salmonids by 1) reducing prey refuge habitat by modifying the shoreline habitats that are critical in all predator-prey interactions; 2) providing concealment structures for ambush predators such as bass and sculpin; 3) creating enough artificial structure to reduce bass home range sizes; 4) providing artificial lighting that allows for around-the-clock foraging by predators; 5) potentially increasing migration routes for smolts and rearing fry, thus increasing exposure to predators; and 6) potentially increasing the bass population by increasing the amount of potential spawning habitat (Kahler et al. 2000).

Docks and piers are often associated with boat traffic. Boating impacts include impacts to submerged aquatic vegetation from prop wash, contamination from fuel discharges, erosion of shoreline due to increased wave action, and resuspension of bottom sediments and turbidity (Kelty and Bliven 2003). Docked boats can also increase light attenuation under the dock or pier, increase turbidity and physical disturbance from propeller wash, scour, and scarring if the propeller hits the substrate (Haas et al. 2002). Water quality impacts such as frequent exposure to petroleum, household cleaners, pesticide products as well as sewage increases with boat usage around docks and piers (Williams et al. 2003a).

Conservation Measures

To minimize, reduce or avoid overwater structure impacts, conservation measures will be implemented during overwater structure repair or replacement. These conservation measures include the following:

- Minimize/reduce pier and overall footprint of structure to reduce shading impacts
- Grating will be installed on more than 50% of the structure
- In marine waters, all piers and floats should be at least 4 feet (1.2 m) above marine vegetation at the MLLW elevation.

See Table 7-1 for corresponding construction methods and conservation measures for the effects of overhead structures.

7.1.1.7 Effects of Vactoring and Excavation

The potential mechanisms by which vactoring and excavation could affect listed fish species include direct mortality, injury by entrainment, sublethal effects (stress, gill damage, and increased susceptibility to disease), and behavioral responses (disruptions to feeding or migration) (Pacific International Engineering 2001). Long-term ecosystem effects of vactoring and excavation generally include changes in the volume and area of habitat, periodic changes to primary and secondary production (food web effects), and changes in hydrodynamics and sedimentology (Nightingale and Simenstad 2001b).

The following are biological effects to listed fish species from vactoring and excavation:

1. Temporary reductions in water quality from suspended sediment associated with vactoring and excavation could reduce or preclude foraging in the affected area
2. Temporary loss of benthic organisms and other prey due to disturbance of the sediment substrates
3. Potential exposure to contaminated sediments or water.

Water Quality

Vactoring and excavation will only occur within streams that are dewatered before removing any sediment. Therefore, vactoring and excavation will not impact water quality.

Within Puget Sound, fine sediment removal will create a sediment plume that may not disperse rapidly because of tidal fluctuations, especially during incoming tides. This could create poor water quality (i.e., decreased dissolved oxygen levels) that might preclude listed fish from accessing foraging and rearing habitat. Excavating activities disturb and suspend sediment, discoloring the water, reducing light penetration and visibility, and changing the chemical characteristics of the water. The size of the sediment particles and tidal currents are typically correlated with the duration of sediment suspension in the water column. Larger particles, such as sand and gravel, settle rapidly, but silt and very fine sediment may be suspended for several hours. Lasalle (1988) described a downstream plume that extended 900 feet (274 m) at the surface and 1,500 feet (457 m) at the bottom. Lasalle (1988) also noted a 70% increase in sediment levels as the bucket descended through the water.

Excavating effects on water quality (suspended sediments and chemical composition) can hurt salmonids. Suspended sediments can have an adverse effect on migratory and social behavior as well as foraging opportunities (Bisson and Bilby 1982, Sigler et al. 1984, Berg and Northcote 1985). Servizi (1988) observed an increase in sensitive biochemical stress indicators and an increase in gill flaring when salmonids were exposed to high levels of turbidity. Gill flaring allows the fish to create sudden changes in buccal cavity pressure, which is similar to a cough (see section *7.1.1.2 Effects of Sediment* above).

Chemical composition of the water with suspended sediments is also affected by excavating activities. Estuarine sediments are typically anaerobic and create an oxygen demand when suspended in the water column, which in turn decreases dissolved oxygen levels (Hicks et al. 1991, Morton 1976). A review of the processes associated with dissolved oxygen reduction (Lunz and LaSalle 1986, Lunz et al. 1988) suggested that dissolved oxygen demand of suspended sediment is a function of the amount of material placed into the water, the oxygen demand of the sediment, and the duration of suspension. The dissolved oxygen reductions appear to be most severe lower in the water column, and usually the condition reverses with adequate tidal flushing (LaSalle 1988). Most research to date indicates that excavating-induced dissolved oxygen reductions are short-term phenomena and do not cause long-term problems in most estuarine systems (Slotta et al. 1974, Smith et al. 1976, Markey and Putnam 1976).

Decreases in dissolved oxygen levels have been shown to affect swimming performance levels in salmonids (Bjornn and Reiser 1991). The decrease of swimming performance due to decreases in dissolved oxygen can be expected to affect the ability of salmonids to escape potential predation or could affect its ability to forage on motile fish. Lasalle (1988) found a decrease in dissolved oxygen levels from 16% to 83% in the mid- to upper water column and nearly 100% close to the bottom. Smith et al. (1976) found dissolved oxygen levels below 2.9 mg/l during excavating activities in Grays Harbor. Hicks (1999) observed salmon avoidance reactions when dissolved oxygen levels dropped below 5.5 mg/l.

Excavating can be conducted using mechanical equipment such as a barge-mounted crane fitted with a clamshell bucket or with an environmental bucket. An environmental bucket, which closes, vents and seals the bucket from leaking, causes very limited, short-term localized turbidity. No long-term effects would result from this turbidity.

Benthic Organisms

Vactoring and excavation will disrupt benthic habitat, temporarily eliminating benthic organisms and will reduce foraging opportunities for listed fish species. This may cause fish to migrate into deeper waters where there is greater vulnerability to predation or into habitat where there are fewer foraging opportunities.

Disruption of the channel bottom and entrainment by vactoring or excavation has a negative impact on benthic biota and forage fish. Removal of sediment in a stream physically disturbs the channel bottom, eliminating or displacing established benthic communities, thus reducing prey availability to salmonids or their forage species. Filter-feeding benthic organisms can suffer from clogged feeding structures, reduced feeding efficiency, and increased stress levels (Hynes 1970). Sediment removal may also suppress the ability of some benthic species to colonize a vactored or excavated area, thus resulting in loss of benthic diversity and food sources for prey species.

Contaminants

Sediment removal within Elliot Bay, Duwamish Waterway, and Lake Washington has the potential for short-term suspension of chemicals if excavation occurs in contaminated sediments. Very little information is known about the toxicity of contaminants to listed fish species. Preliminary work with freshwater toxicity levels indicates that they are sensitive to contaminants. Hansen et al. (2000) found effects to bull trout from cadmium as low as 0.089 µg/L, which is much lower than EPA's chronic water quality criterion of 0.9 µg/L. Collier et al. (2000) suggest that current sediment quality criteria (established by EPA) for PCBs, TBT, and PAHs for juvenile salmonids may be inadequate to prevent damaging their disease resistance, causing DNA damage, or reducing their prey base. Ongoing research by Hansen et al. (2000) has shown that measured LC50s for bull trout from cadmium and zinc were less than the national water quality criteria. Cook et al. (1999) demonstrated that bull trout were 3 times more sensitive to certain contaminants than lake trout using egg dose-dependent mortality data to 2,3,7,8-tetrachlorodibenzo-p-dioxin and PCBs. Although preliminary, most of the bull trout toxicity work has concluded there are effects to bull trout at levels lower than the existing water quality standards, and bull trout will be impacted by increases in contaminant levels in the water column. Other effects of contaminants to listed fish are described above in section *7.1.1.5 Effects of Pile Removal*.

Conservation Measures

Conservation measures for vactoring, and excavation are those that minimize sediment input into the stream (i.e., TESC plan, minimizing heavy equipment and stream crossing sedimentation) and habitat degradation. See Table 7-1 for corresponding construction methods and conservation measures for the effects of vactoring and excavation.

7.1.1.8 Effects of Shoreline Hardening, Bank Stabilization, and Habitat Enhancement and Restoration Activities

Shoreline Hardening: Bulkheads

Bulkheads can have a variety of impacts on the aquatic environment due to construction, maintenance, or existence (Kahler et al. 2000). Some of these effects include:

- Temporary increases in turbidity associated with construction
- Disruption of migratory and rearing behavior of juvenile salmonids
- Removal of vegetation
- Reduction or elimination of sediment recruitment to the lake or shoreline
- Elimination of shallow-water habitat
- Reflection of wave energy along the shoreline that increases scour of sediment
- Permanent removal of woody debris.

These impacts result in numerous effects to salmonids including reduced prey abundance, decreased habitat complexity, decreased shallow water, increased predation, increased chemical contaminates, and increased high energy environment (Kahler et al. 2000). Williams and Thom (2001) state that possibly the most significant effect of hardened shoreline stabilization is a direct impact to regional geomorphology via impoundment of potential natural sediment sources (Macdonald et al. 1994). Structures located above the natural beach grade can cut off sediment supply from a feeder bluff or upper beach. They will cause direct onsite impacts to habitat structure (e.g., shift to a lower elevation, higher energy, hard substrate shoreline), as well as indirect impacts within the coastal drift cells (Downing 1983).

The placement of hardened structures along natural shorelines can influence erosion processes that alter the structure and function of native habitats at areas both near and far from site of impact. This effect appears to be consistent throughout protected bay and estuarine habitats, as well as outer coast environments. For example, in a field survey of the entire developed ocean coasts of South Carolina, North Carolina, and New Jersey, Pilkey and Wright (1988) showed that dry beach width was significantly narrower in front of stabilized seawalls and that areas with a higher degree of stabilization correlated to narrower beaches. Limited quantitative understanding of interactions between shoreline processes and hardening structures continues to fuel debate over the cumulative effects of shoreline armoring on beaches and adjacent properties (Pilkey and Wright 1988). However, most evidence suggests biological communities do respond locally to physical changes.

Structural modifications may directly alter shoreline geomorphology including tidal elevation relative to MLLW, gradient, channel characteristics (depth, width, crosssectional area, sinuosity), and sediment character and quality. Geomorphology affects rates of tidal inundation and exchange, and is responsible for most of the distinguishing physical and chemical features of tidal systems. Placement of structures below the OHW mark often results in a permanent loss of habitat, reducing the availability and extent of intertidal foraging, spawning, and refuge areas. Changes in the physical composition and volume of substrates have predictable effects on biological resources (Macdonald et al. 1994, Dethier 1990, Thom et al. 1994). Long-term, chronic impacts may reduce intertidal habitat area, bottom complexity, and associated soft-bottom plant and animal communities.

Hardened shorelines with vertical or recurved slopes (like rock jetties) alter hydrology by deflecting wave energy downward, scouring the bottom sediment at the toe and periphery (Engineering Science 1981, Zabawa and Ostrom 1982). This ultimately results in elevation loss and habitat change. Added turbulence and scour may prevent vegetation establishment and alter the floral assemblage (Watts 1987, Thom 2002). Loss of sediment supply can erode beach profiles and lower the beach gradient. This change will result in loss or impairment of species and communities adapted for using higher elevations and particular substrates.

Hardened shorelines built below the MHHW line can steepen the natural shoreline, an effect created by the steep face of the structure, and can eventually, after several years, result in an increase in the mean water depth and a corresponding loss of the shallow, intertidal habitat preferred by juvenile salmonids as a migration and foraging corridor (Douglass and Pickel 1999). During periods of high tide, the water along the submerged face of the bulkhead will be deeper, with a steeper slope, than the shallow-water habitat found along a natural, gradually sloping beach.

Over time, shoreline hardening is expected to alter the physical characteristics of beach and nearshore biotic communities. These changes in turn can alter distribution and abundance of fish within the action area.

Bank Stabilization

Bank stabilization techniques in a dynamic river environment reduce the potential for channel complexity by limiting channel migration and recruitment of large woody debris and gravel. Rivers continuously transport eroded material downstream from areas of erosion to areas of deposition. Transport varies with discharge and is therefore episodic (Kondolf 1994). Armoring streambanks limits lateral channel changes and gravel recruitment (Schmetterling et al. 2001).

Bank hardening may also sequester onsite gravel sources from capture by the active river system and cause downcutting due to increased flow velocities. Downcutting may extend well upstream or downstream, and result in the perching of historic depositional gravel layers above the OHW, thereby reducing gravel capture rates within the system.

A net loss of gravel recruitment to the system may ultimately result in the loss of sufficient gravels to support successful salmon spawning. The cumulative effect of gravel isolation may lead to the loss of enough sources that the waterway becomes gravel-limited. Overall, streambank stabilization will reduce the potential for side channel formation and lateral channel migration in the floodplain, which are natural processes contributing to habitat complexity. These processes contribute to undercut banks and overhead cover that help provide important summer habitat for salmonids (Brusven et al. 1986, Beamer and Henderson 1998).

The placement of riprap above and below the OHW will permanently degrade the streambed substrate in streams within the action area. Placement of riprap on top of the streambed may injure or kill Puget Sound Chinook salmon, bull trout, and/or steelhead juveniles that hide in interstitial spaces. Riprap installation results in the following:

- Removal of native sediments
- Installation of different sized sediments (riprap)
- Reconstruction (stabilization) of the streambank slope.

Such activities can be characterized as channelization. Bolton and Shellberg (2001) describe channelization as the deliberate or indeliberate alteration of 1 or more of the interdependent hydraulic variables of slope, width, depth, roughness or size of sediment load. Thus the effects of the habitat alteration related to the installation of riprap can be evaluated as channelization.

Channelization has immediate and direct effects on stream processes because it involves direct modification of the river channel. These effects result in both physical and biological changes that lead to various alterations of biological systems. The changes affect benthic macroinvertebrates, fish, and aquatic riparian vegetation from algae and macrophytes to riparian shrubs and trees.

A typical sequence of events that occurs after the placement of a channelization activity leads to immediate changes in physical aspects of the channel. These physical changes lead to longer-term biotic responses that extend over space and time (Simpson et al. 1982 in Bolton and Shellberg 2001). The biological effects may be in response to the physical changes in depth, shade, sediment temperature, altered hydrology, isolation of floodplain habitats, etc. Or they may be in response to changes in nutrient cycling and changes in population of various trophic (nutrition) levels that get transmitted throughout a biological system. Streamflow, stream velocity, channel morphology, vegetation and channel substrate are all affected by channelization activities. The physical nature of stream channels reflects a continuous readjustment of the interrelated variables of discharge, slope, channel width and depth, flow velocity, channel roughness and sediment characteristics (Brookes 1988).

Some studies have looked at the biological effect of specific structures and bank stabilization techniques, such as riprap, spur dikes, and revetments. Hjort et al. (1984) looked at fish and invertebrates along revetments and natural channel areas of the Willamette River, Oregon. They found different numbers and species of fish and invertebrates in natural stream areas compared with riprap banks. Fewer fish species used riprap areas than used natural areas. Fish found in revetment areas tended to be ones that fed on algae or diatoms growing on the stones or fed on bottom-dwelling invertebrates. Invertebrates found in the revetments were species that preferred a very stable bottom and either clung to stones or hid in crevices. More fish species were found in areas with natural banks due to the greater diversity of habitat in these areas.

Li et al. (1984) compared larval, juvenile and adult fish use of natural and channelized habitats in the Willamette River, Oregon. They concluded that continuous revetments are not good larval fish habitat. The combination of proximity to fast water, steep bank slopes, greater water depth, and cooler temperatures does not provide suitable habitat for larval fish. Spur dikes have a greater diversity of habitats than continuous revetments and appear to be intermediate in habitat quality between natural banks and continuous revetments. Low-angle beaches that develop between spur dikes can provide good larval fish habitat. Natural banks have the greatest diversity of habitats within secondary channels, fast and slack water areas and backwaters. And, as expected, natural banks have the most diverse fish species composition.

Peters et al. (1998) looked at seasonal fish densities in Washington at sites with various bank stabilization structures. They conducted a survey of typical bank stabilization methods and found that 496 of 667 projects used riprap or riprap with deflectors. Only 29 projects used bioengineering or large woody debris. Of all project types (riprap, riprap with large woody debris, rock deflectors, rock deflectors with large woody debris and large woody debris) they surveyed, only sites stabilized with large woody debris consistently had higher fish densities in spring, summer and winter than the control sites without any stabilization structures. Riprap sites consistently had lower densities than control sites. At all sites, fish densities were generally positively correlated with increasing surface of large woody debris and increasing amounts of overhead riparian cover with 12 inches (30 cm) of the water surface.

The effects of streambank alteration are not limited to the wetted stream channel itself. Connectivity longitudinally (up and downstream), laterally (floodplain and uplands) and vertically (groundwater, hyporheic, and phreatic) is a major feature of stream corridors (Stanford and Ward 1992). The temporal nature of the system adds a fourth dimension (Ward 1989). These linkages mean that the effects of channelization can be transmitted over areas far beyond an actual work zone. Impacts include changes in hydrology, biology, morphology, and water quality (Brookes 1988).

Lateral connectivity is altered by channelization activities including dredging and filling, channel lining, and bank stabilization. The cessation of overbank flooding and the flood-pulse (Junk et al. 1989) effect is suspected to decrease floodplain productivity and biodiversity (Bayley 1995).

Longitudinally, connectivity is most clearly affected by diversion structures that either store or remove water, sediment, and nutrients from the river (Ward and Stanford 1995). Diversions can have a significant effect on the quantity and timing of flow in the river, water temperature, and sediment and nutrient loads (e.g., Lillehammer and Saltveit 1984, Ligon et al. 1995).

Many observations indicate that downstream flooding is a common—but not inevitable—response to channelization. If the channelization decouples the timing of peak flows merging at confluences, downstream flooding may be decreased. Draining and filling of wetlands and swamps in floodplains reduces the storage capacity of the system and leads to more downstream flooding (Brookes 1988).

Onsite effects of channelization typically increase channel slope and water velocity. As a result, more sediment is eroded and transported downstream where it is deposited in areas that have not had transport capacity altered. Morphologically, this leads to incision or widening of the channel onsite and aggradation (filling) of the channel downstream when the sediment is deposited.

Water quality effects are highly site-specific. They are controlled by watershed land use, extent of channelization, and length of the recovery period (Brookes 1988). Shields and Sanders (1986) reviewed studies on the effects of excavation and diversion on water quality. They found water quality changes were due to increased sediment inputs and decreased shade. Most of the measured water quality parameters increased by 50% to 100% during construction compared with pre-construction values. Little (1973) reported that during and after channelization, large amounts of suspended sediments are typically released and deposited downstream where they adversely affect aquatic life. If the channelized reach is very long, reduced shade may increase temperatures downstream (Duvel et al. 1976). Few studies have directly addressed the effects of channelization on water quality components such as oxygen, nutrients, and ions (Brookes 1988).

Typically, changes due to human activities in the channel migration zone reduce habitat diversity, which affects the numbers and kinds of animals the habitat can sustain (Schneberger and Funk 1972, Hahn 1982, Simpson et al. 1982). As the physical habitat changes, stresses are placed on individual plants and animals. These stresses—depending on the tolerance of the species and individual—may limit growth, abundance, reproduction, and survival (Lynch et al. 1977). Biologically important parameters that change following channel activities include water temperature, turbidity, flow velocity, variable water depths, hydrologic regime, a decrease or change in vegetation, changes in storage of organic matter and sediment, and changes in the size and stability of channel substrate (Hahn 1982). These changes can decrease habitat connectivity and the exchange of energy and matter between habitats. The direction of change varies by site and circumstance. Specific structures proposed to be installed and potential impacts to listed fish are shown in Table 7-6:

Table 7-6
Typical structures for bank stabilization

Structure	Function	Effect
Groins and/or barbs	Roughness elements that extend from the bank into the water to direct flow away from an eroding bank. Groins and barbs are similar except groins are larger and tend to deepen and narrow the stream.	Groins and barbs direct water away from one side of a stream to the opposite side which can increase bank erosion, thus increasing the need for additional bank stabilization methods.
Drop structures and porous weirs	Low-elevation weirs that span the entire width of the channel designed to spill and direct flow away from an eroding bank, dissipate energy and provide grade stabilization. Drop structures are not porous and are usually constructed with logs or concrete.	Drop structures not installed correctly may result in increased scour downstream of the structure that may create a fish passage barrier. A fish barrier may also result if the upstream-to-downstream water surface elevation is excessive.
Log toes	Structural features that prevent erosion at the toe of a streambank. Log and rootwad toes provide a natural approach to toe protection.	They are very effective at controlling bank erosion, but can also increase water velocities that can result in further downstream erosion. As with most hardened bank structures, log toes result in lost opportunities for sediment supply and recruitment of large woody debris.
Coir logs	Long, sausage-shaped bundles of coir (coconut fiber), bound together with additional coir or synthetic netting. They provide biodegradable stabilization to streambanks.	They decompose over 7 to 12 years and provide good moisture-retention properties. Coir logs are also placed on top of streambanks on exposed soils to control sediment input into streams.
Riprap	Bank armoring consisting of rock for controlling bank erosion. Riprap is very effective at controlling bank erosion but results in a permanent lost opportunity for sediment and large woody debris recruitment.	Riprap has very little aquatic-habitat value or cumulative effect on channel forming processes. Riprap tends to increase water velocities downstream, which results in increased bank protection measures.
Woody planting	Placement of woody planting to stabilize eroding banks, provide habitat benefits and improve aesthetics.	Woody plantings are not very effective during their first growing season. They provide structural habitat diversity to banks and can provide overhanging cover for fish.

Habitat Enhancement and Restoration Activities

Large Woody Debris. Installing large woody debris into bank stabilization and habitat enhancement and restoration project designs will provide shade, cover, and contribute to habitat complexity. Large woody debris is central to determining channel morphology and biological condition in many Pacific Northwest streams (Spence et al. 1996). Pool formation, gravel and organic material retention, velocity disruption, and cover for fish from predators are all strongly reliant on large woody debris. Other than natural mortality, sources of large woody debris recruitment to streams include bank erosion, blow down, and transport from upstream (Gurnell et al. 1995). The replanting of native vegetation provides a future source of large woody debris recruitment.

Boulders and Boulder Clusters. Boulders and boulder clusters increase and restore structural complexity, hydraulic diversity, and fish habitat. Placement of boulders and boulder clusters creates a diversity of water depth, substrate, and velocity. Boulders confine and direct flow, creating bed and bank scour and depositing sorted bed material that provides cover and spawning habitat (WDFW 2004).

Depending on the design, spacing, and location of boulders, they may have a backwater effect on the upstream reach of the channel. This backwater effect can cause upstream deposition, and possible increase in a floodwater state. If not properly designed and installed, increased bank erosion may occur.

Boulder placements typically pose a low risk to existing habitat. Potential impacts would include temporary loss of habitat value associated with sediment movement and depositions through scour and slower water velocities. If upstream backwater effects occur resulting in sediment deposition, sediment may need to be excavated to obtain the desired effects of boulder installation.

Weirs or Groins. Groins are large roughness elements that project into the channel of a stream from the bank and extend above the high-flow, water-surface elevation. The main function of a groin is to redirect flow away from a streambank to reduce flow velocities near the bank to increase sediment deposition. Barbs are similar to groins except they are not as high profile and have less effect on the cross-section shape of the stream (WDFW 2003).

Weirs are low-elevation structures that span the entire width of the stream channel. Two main types of weirs are 1) drop structures and 2) porous rock weirs. Drop structures are designed to create substantial more backwater. They can be constructed with rock, logs, sheet piles, or concrete. Porous weirs are constructed of loosely arranged boulders that redirect flows away from the bank and toward the center of the channel.

Groins and barbs constrict the channel by blocking a portion of the channel. This can increase erosion on the opposite bank as the water is pushed toward that side of the stream. Groins and barbs also push the thalweg of the stream away from the bank. This may result in downstream channel adjustment and increased erosion of the stream substrate or banks. Groins and barbs prevent channel migration, which reduces sediment and large woody debris recruitment into the stream. Existing spawning habitat may be lost due to increased erosional forces as the channel is constricted and the thalweg is pushed away from the bank. Incorporating large woody debris into groins and barbs will minimize these effects.

Drop structures are designed to spill and direct flow away from an eroding bank, dissipate and redistribute energy, and provide grade stabilization. Drop structures constrict flows to a specific location in the channel that creates a scour hole, plunge pool at the constriction point. If not properly installed, a fish barrier may result from the difference in surface elevations. Existing spawning habitat may be lost due to installation of drop structures, but some spawning habitat may be formed by sediment deposition at the downstream portion of the plunge pool.

Porous weirs are similar to drop structures but are not as rigid and are designed to have spaces between the boulders to allow fish and sediment to pass through the structure. Porous weirs are designed to redirect flow away from the bank and to provide channel roughness. Redirection of flow is caused by constricting flow between boulders, which increases erosive forces downstream and sediment transport. Porous weirs may affect spawning habitat similarly to drop structures.

Conservation Measures

Numerous conservation measures will be incorporated into shoreline and nearshore habitat modification and bank stabilization projects. Conservation measures incorporated into projects are intended to create salmonid and/or prey species habitat or decrease hard bank and shoreline structures. The main conservation measures include:

- Reduce sediment input into the stream
- Avoid fuel/oil contamination of the site from equipment operation
- Reduce bulkhead impacts by removing the bulkheads from the water and installing them behind the OHW or the MHHW line.
- Increasing habitat complexity around the bulkheads with large woody debris, cove installation, and riparian vegetation.
- Increasing habitat complexity in riprap by including large woody debris, and filling interstitial spaces with habitat mix.

See Table 7-1 for the construction methods and conservation measures for the effects of shoreline hardening, bank stabilization and habitat enhancement and restoration.

7.1.1.9 Culvert Replacement

The overall impact of a proposed culvert project on listed fish species is expected to be beneficial because it will restore spatial and temporal connectivity of waterways within and between watersheds where movement is currently obstructed. Connectivity will permit listed fish species to access areas critical for fulfilling life-history requirements, especially foraging, spawning and rearing.

The constricted flows at culverts or bridges are largely due to poor installation or undersized structures. In many instances high water velocities amplified by undersized culverts have created large scour pools at the culvert discharge point, altering the stream elevation below the natural gradient. Over time, culverts become elevated above the stream and create a physical barrier to fish passage. In other cases, water also drains under and around culverts, and migrating fish attempting to follow these flows can become stranded or impinged against the culvert or road fill.

In addition to allowing for fish passage for all age classes, the replacement or removal of fish-blocking culverts should result in more naturally maintained stream hydraulics, including bedload movement, sediment transport, and passage of moderately-sized woody debris, leading to more natural stream dynamics and stream geometry. The new structures should result in fewer maintenance needs and better performance during high precipitation events, resulting in near-normal sediment and bedload movement and debris conveyance.

Each culvert replacement will also include restoration of the streambed within and immediately downstream and upstream of the culvert. Stream restoration will include the placement of large woody debris, boulders, and spawning gravels with the goal of increasing habitat complexity of the aquatic environment currently lacking at many culvert sites. Placement of these materials should aid in improving the habitat value for listed fish species and their prey species.

With the onset of fish removal and construction activities, listed fish species will experience short-term adverse effects due to fish removal and relocation procedures before or along with stream dewatering and isolation of the project work area. This will disrupt normal fish behavior and in some instances, cause mortality. Construction impacts will have localized effects to the riparian corridor. The effects of sediment to the aquatic environment during construction are expected to be minimal due to the construction occurring in dewatered streams and other sediment control measures being implemented at each construction site. However, rain during and after construction will likely mobilize sediment into the stream, even with sediment control measures in place, because those measures are not always effective at precluding sediment deposition into streams (Rashin et al. 1999).

Sedimentation and turbidity will occur from heavy equipment operation on access roads and excavation/fill areas by exposing, destabilizing, and/or compacting streambanks, streambeds, and riparian soils. Access roads will be built from the existing road to the stream in a direct line to the stream diversion and discharge point or to the structure, as needed. Heavy equipment operation in streambeds will only occur during dewatered periods. Additional sedimentation may occur from excavating the roadfill (above the wetted perimeter), backfilling, clearing and restoring the riparian area, maintenance, and repairing streambeds following high-flow events.

After construction, periodic spikes in sediment input are expected during the first winter season in response to precipitation events that may mobilize unstable sediments from upland locations. Sedimentation may also occur throughout the site recovery period until fill slopes stabilize.

7.1.1.10 Effects of Boating Activity

Adding or improving boat launches, docks, and piers may increase levels of boating activity. Boating activities can cause several impacts on listed salmonids and aquatic habitat. For example, the following can occur with boating (Mueller 1980, Warrington 1999):

- Engine noise
- Prop movement
- Physical presence of boat hulls may disturb or displace nearby fish.

Boat traffic increases the following:

- Turbidity and up-rooting of aquatic plants in shallow waters
- Aquatic pollution (through exhaust, fuel spills, or release of petroleum lubricants)
- Shoreline erosion.

These boating impacts affect listed fish several ways. Turbidity may injure or stress fish. The loss of aquatic macrophytes may expose salmonids to predation, decrease littoral productivity, or alter local species assemblages and trophic interactions. Despite a general lack of data specifically for salmonids, pollution from boats is thought to potentially cause short-term injury, physiological stress, decreased reproductive success, cancer, or death. Further, pollution may also affect fish by impacting potential prey species or aquatic vegetation. Shoreline erosion can change hydraulic flow patterns, increase sedimentation and turbidity, reduce riparian vegetation, and steepen bank and nearshore gradient.

See Table 7-1 for construction methods and conservation measures for the effects of boating activity.

7.1.1.11 Effects of Pesticides

While there is a healthy volume of literature regarding pesticide effects to aquatic species, in some cases, data are lacking for a specific pesticide on particular salmonid species and their prey, including diverse life-stages. ‘Pesticides’ in this document refer to all chemicals used to control unwanted insects (insecticides), weeds (herbicides), or other activity such as killing roots in pipes. No chemical fertilizers are used to establish plant restoration.

Pesticide Application

The application of pesticides in proximity to lake and river systems can result in the transport of potentially toxic chemicals (active ingredients or adjuvants) to surface waters (USGS 1999) that may harm ESA-listed species. Pesticides can impair the essential biological requirements of salmonids if they undermine the physical, chemical, or biological processes that collectively support a productive aquatic ecosystem (Preston 2002) or affect the physiological or behavioral performance of salmonids in ways that will reduce growth, survival, migratory success, or reproduction.

The degree, or likelihood, of effects to ESA-listed salmonids from the discharge of pesticides to surface waters vary spatially and temporally, according to factors that have been simplified into the following categories:

1. Likelihood of Exposure. If listed fish do not occupy habitat that has been chemically modified, the likelihood of effects could be limited to loss of prey base.
2. Water Quality Conditions. Dissolved oxygen levels and temperature affect salmonids susceptibility to pesticide exposure.
3. Lifestage of the Salmonid. Salmonids occupy freshwater as incubating eggs/alevins, newly emerged fry, and rearing parr and smolts, and as returning adults. Each lifestage has a different susceptibility or tolerance of exposure to pesticides.
4. Levels of other Contaminants. Concurrent discharge or background levels of other contaminants can magnify effects through mixture toxicity resulting from discharges associated with the use of the chemical.
5. Concentration and relative toxicity of the chemical.

Pesticides can impair the essential biological requirements of salmonids if they undermine the physical, chemical, or biological processes that collectively support a productive aquatic ecosystem (Preston 2002). The alteration of watershed characteristics by pesticides can include: 1) disruption of the growth of riparian deciduous vegetation, 2) reduction of delivery of leaves and intermediate-sized wood, and 3) alteration of hydrologic and sediment delivery processes (Spence et al. 1996). Moreover, aquatic plants and macroinvertebrates are generally more sensitive than fish to the toxic effects of pesticides. The application of pesticides can affect the productivity of the stream by altering the composition of benthic algal communities, the food source of macro-invertebrates. Benthic algae are important primary producers in aquatic habitats, and are thought to be the principal source of energy in many mid-sized streams (Minshall 1978, Vannote et al. 1980, Murphy, 1998). Pesticides, specifically herbicides, can directly kill algal populations at acute levels or indirectly promote algal production by increasing solar radiation reaching streams by disruption of riparian vegetative growth. The disruption of riparian vegetative growth carries with it other consequences for salmonid habitat, such as loss of shade, bank destabilization, and sediment control. Therefore, pesticides can potentially impact the structure of aquatic communities at concentrations that fall below the threshold for direct impairment in salmonids. The integrity of the aquatic foodchain is an essential biological requirement for salmonids, and the possibility exists that pesticide applications will alter the productivity and watershed characteristics of streams and rivers.

Pesticides can cause significant shifts in the composition of benthic algal communities at concentrations in the low parts per billion (Hoagland et al. 1996). Based on the data available, pesticides have a high potential to elicit significant effects on aquatic microorganisms at environmentally relevant concentrations (DeLorenzo et al. 2001). In many cases however, the acute sensitivities of algal species to pesticides are not known. In addition, Hoagland et al. (1996) identify key uncertainties in the following areas: 1) the importance of environmental modifying factors such as light, temperature, pH, and nutrients, 2) interactive effects of pesticides where they occur as mixtures, 3) indirect community-level effects, 4) specific modes of action, 5) mechanisms of community and species recovery, and 6) mechanisms of tolerance by some taxa to some chemicals. Pesticide applications have the potential to impair autochthonous (indigenous) production and, by extension, undermine the trophic (food) support for stream ecosystems.

Prey Base Effects and Bioaccumulation

It is becoming increasingly evident that the indirect effects of contaminants on ecosystem structure and function are a key factor in determining a toxicant’s cumulative risk to aquatic organisms (Preston 2002). Adverse effects to salmonid prey base can occur from exposure to some substances. Aquatic plants and macroinvertebrates are generally more sensitive than fish to the acutely toxic effects of pesticides. Therefore, chemicals can potentially impact the structure of aquatic communities at concentrations that fall below the threshold for direct biological impairment in salmonids. The integrity of the aquatic foodchain is an essential biological requirement for salmonids, and the reasonable likelihood pesticide applications will reduce the productivity of streams and rivers is a significant effect.

Pesticide effects to salmonid prey base typically occur through 2 primary mechanisms: 1) effects to the amount and/or type of food supply, or 2) by exposure via food organisms. Depending on the exposure scenario, effects to aquatic invertebrate communities can be very short-term, or take months or years to fully recover. Exposure via food organisms is likely to be much more episodic and short-term. Norris et al. (1991) provide a summary and literature review of pesticide effects to salmonids. The amount and/or type of food supply can be altered by pesticides in complex and subtle ways, particularly if the aquatic system is exposed to a combination of pesticides.

Pesticides can alter the prey base by direct mortality of aquatic invertebrates (Beschta et al. 1995). Pesticides can cause direct mortality of aquatic invertebrates, or trigger extensive drift of aquatic invertebrates out of the affected area (Spence et al. 1996). If grazing invertebrates are reduced or eliminated from a stream reach, primary production release may occur (such as algal blooms), altering trophic structure.

Pesticides are often not highly toxic to salmonids, as they are generally designed to interfere with physiological systems unique to plants. However, low concentrations of pesticides may exert significant effects on salmonid prey items by affecting algal or aquatic plant communities (Pratt et al. 1997), or directly on salmonids through sublethal effects of the pesticide (Spence et al. 1996). In addition, some pesticides, such as triclopyr esterare, are moderate to highly toxic to aquatic invertebrates (SERA 2003), and adjuvants and surfactants present in pesticide commercial formulations can greatly enhance toxicity (SERA 1997, Stark and Walthall 2003).

Salmonid pesticide exposure through food organisms can occur through incidental exposure of terrestrial insects that subsequently become prey items for fish (Norris et al. 1991), or indirectly through invertebrate ingestion of organic material delivered to the aquatic system (Urban and Cook 1986). Pesticides that are more lipophilic (fat soluble) will tend to partition into organic material in or on soil. Runoff can mobilize organic material into streams where it is consumed by insects and crustaceans. Little data are available on the risk of exposure via this pathway, but risk is likely to be highly variable depending on conditions at the time of application, such as seasonal timing.

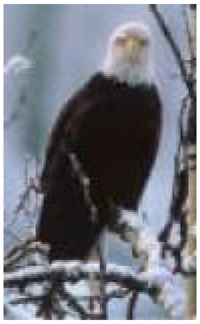
Bioaccumulation in fish is partially mediated by the presence of pesticides in food items and sediment residues, but also includes bioconcentration, defined as passive uptake from the water column (Klaassen et al. 1986). The lipophilicity of the pesticide and fat content of the organism are the primary factors determining the extent of bioaccumulation. Pesticides with high lipophilicity tend to partition out of the water column and into food items, with the degree of partitioning proportional to the organism fat content. Concentration up the foodchain (biomagnification) occurs when repeated exposure through consumption of contaminated prey items results in high concentrations of pesticides in predators, such as salmonids. For bioaccumulation to occur, a pesticide must have sufficient lipophilicity and persistence, and relatively low acute toxicity.

The possibility exists of effects from additive, antagonistic or synergistic effects from multiple applications. The relative risk of these types of effects depends on the volume and timing of their delivery, and background water quality conditions. Within the zones of possible exposure periods described above, the greatest likelihood of additive/synergistic effects from applications would occur anytime precipitation causes significant subsurface or overland flow delivery to aquatic systems. The volume and types of pesticides delivered would depend on the relative success of the pesticide to inhibit off-target delivery. As precipitation levels rise, subsurface and overland flow will increase, thus pesticide delivery to nearby streams is reasonably likely to occur.

Conservation Measures

Conservation measures included during pesticide application are intended to minimize improper application. A licensed applicator must oversee that pesticides are being applied properly. In addition, pesticides must be used for the intended purpose of killing, removing, or controlling unwanted species. See Table 7-1 for corresponding construction methods and conservation measures for the effects of pesticides.

7.1.2 Bald Eagles



7.1.2.1 Effects on Nesting Eagles

Disturbance

Nesting territories within an action area are subject to disturbance from construction and potential long-term project use. Any potentially disturbing activity in excess or under the right conditions can alter a bald eagle's normal behavior or induce nesting failure (Grubb and King 1991). The response of nesting eagles to human activity can range from behavioral, such as flushing or reduced nest attendance, to nest failure (Fraser et al. 1985, McGarigal et al. 1991, Grubb and King 1991, Grubb et al. 1992, Anthony et al. 1995, Steidl and Anthony 1996, Watson and Pierce 1998). The magnitude of the response varies inversely with distance and increases with disturbance duration, the number of vehicles or pedestrians per event, visibility, sound, and position relative to affected eagle (Grubb and King 1991).

Bald eagles vary in their sensitivity to disturbance, but generally nest away from human disturbance (Stinson et al. 2001). Watson and Pierce (1998) found that vegetative screening and distance were the 2 most important factors determining the impact of disturbances. Heavy vegetative screening can dramatically reduce eagle response to human activity. Human activities that are distant, of short duration, out of sight, few in number, below the nest, and quiet have the least impact (Grubb and King 1991). Parson (1994) reported that successful nests had lower densities of human residences within about 295 feet (90 m) than unsuccessful nests. Larger set-back distances for buildings have been correlated with greater eagle use. Hodges et al. (1984) reports that in coastal British Columbia, adult eagles and active nests were found in higher than expected numbers in undisturbed habitat, and that disturbed habitat with no remnant old-growth contained far fewer adult birds and no active nests. Grubb et al. (1992) reported the threshold for alert response was about 1,800 feet (549 m) (and for flight response was about 650 feet (195 m) for breeding bald eagles in Michigan and Arizona, with vehicles and pedestrians eliciting the highest response frequencies.

Bald eagle tolerance of disturbance may depend in part on prior experience and the level of the nesting population relative to carrying capacity. Disturbance experiments conducted by Steidl and Anthony (2000) suggested that bald eagles habituated somewhat over 24 hours to camping about 330 feet (100 m) from nests, but the tendency was not cumulative, with each disturbance being essentially independent of the last. Bald eagles exhibit strong year-to-year fidelity to a nest territory and have been shown to be reluctant to abandon a territory despite increased disturbance and habitat alteration (Stinson et al. 2001). A small but apparently growing number of bald eagles in Washington have been exhibiting an unexpected tolerance to human presence and activities, and nesting successfully in close proximity to homes (Watson et al. 1999). However, this may be the result, in part, from a local shortage of nesting habitat. Nest site fidelity may be stronger when the population is at carrying capacity and no vacant suitable sites are available (Stinson et al. 2001).

Bald eagles may be deterred from nesting, perching, foraging, or wintering within 0.25-mile (0.4 km) of project sites if there will be increases in pedestrian and vehicular traffic. An increase in traffic is not anticipated within Seattle because most areas within the City are already highly urbanized. However, an increase in activity due to future projects less than 0.25-mile (0.4 km) from bald eagle nests can affect bald eagle behavior indirectly through the associated increase in pedestrian activity (Watson and Pierce 1998). Studies have shown pedestrian traffic is more disturbing than auto traffic or aircraft (Fraser et al. 1985, Grubb and King 1991, Grubb et al. 1992).

Pile driving generates the highest noise level of all common construction activities (Bolt et al. 1971). Noise measurements of impact driving of steel piles taken by Washington State Ferries at the Anacortes terminal recorded Lmax readings (peak sound emitted from a source) that averaged between 105 to115 dB at 50 feet (15.3 m) (Visconty 2000). Heavy equipment operation for road construction generates noise levels of 77 to 96 dB at 50 feet (15.3 m). A general equation of noise propagation for pulsed sound in air is that there is a 7.5 dB loss for each doubling of distance in areas of soft (forested) ground cover. Noise begins to disturb most birds at 80 to 85 dB, and the sound level threshold for the flight response is around 95 dB (Awbrey and Bowles 1990).

Bottorff et al. (1987) observed bald eagle behavior in response to wood or steel pile driving and determined that impact driving of steel piles may have flushed bald eagles at 4,000 feet (1,219 m). Stanford et al. (1997) determined density and distribution of bald eagles during construction of a dam on the Ohio River and documented a significant reduction in bald eagle numbers within 1 mile (1.6 km) of the construction site. The mean distribution of bald eagles also shifted from a point 0.5-mile (0.8 km) upstream from the dam construction site to a point 1.5 miles (2.4 km) upstream. Pile driving was identified as the most notable disturbance during construction of the dam. Impact driving of steel piles could result in a flight response for any bald eagles within a 1-mile (1.6 km) radius of a project site.

Adequate incubation time and adult perch time near the nest were the best predictors of bald eagle nest success in Washington (Watson and Pierce 1998). Incubation time for bald eagles must be above certain minimum levels and without excessive exposure of eggs in order for embryos to grow and hatch. Exposed eggs weaken the embryos and reduce hatchability (Watson and Pierce 1998). Human or natural events that increase egg exposure by flushing incubating bald eagles for extended periods can cause embryos to die and nests to fail (Watson and Pierce 1998). Disturbance reduces the time bald eagles spend incubating, and decreased incubation time reduces nesting success. Pile driving within 1 mile (1.6 km) and any activity within 656 feet (200 m) of the nest during incubation could cause a flush response, which would reduce incubation time and may affect nest success.

After eggs hatch, Watson (1993) suggested that regular disruption by aircraft or other human activities could result in reduced attentiveness and nest failure due to reduced brooding and feeding of young. In Alaska, humans camping about 330 feet (101 m) from nests for 24 hours caused clear and consistent changes to behavior, including a reduction of 29% in the amount of prey fed to nestlings (Steidl and Anthony 2000). Pile driving within 1 mile (1.6 km) and any human activity that occurs within 656 feet (200 m) of the nest during the nestling period could result in reduced brooding and feeding of young, which could result in nest failure.

Habitat

Assuming the presence of an adequate food supply, the single most critical habitat factor associated with bald eagle nest locations and success is the presence of large super-dominant trees (Watson and Pierce 1998). The average life expectancy of nests is 5 to 20 years. Therefore, bald eagles need trees of similar stature located nearby to serve as replacement nest trees if a nesting territory is to persist (Stinson et al. 2001). Anthony and Isaacs (1989) recommended a 0.25-mile (0.4 km) primary buffer zone around nests to minimize the vulnerability of the nest area to blowdown from wind, fire, disease, and insect infestation. They also recommended against road building, hiking trails, and boat launches less than 0.25 mile (0.4 km) from bald eagle nests, based on their finding that such alterations or the associated human activities were correlated with reduced nest success. Habitat alteration that removes large trees and prevents their replacement could prevent bald eagles from nesting within 0.25-mile (0.4 km) of a project site.

Projects that result in permanent facilities or increased activity will result in increases in both noise and visual disturbance of bald eagles in any adjacent suitable habitat. Fraser et al. (1985) concluded that "Chronic disturbance results in disuse of areas of human activity . . . thus, human activities that chronically exceed the limits of eagle tolerances, may be considered a form of habitat destruction." Passive displacement may impact habitat that otherwise is undegraded. Passive displacement occurs when human use prevents eagles from using a site (Stinson et al. 2001). Passive displacement has not been widely investigated, but may be more prevalent and important than active disturbance that briefly affects birds (McGarigal et al. 1991, Anthony et al. 1995).

Loss of vegetation around the nests could have long-term negative impacts to the nests themselves by reducing protective screening. Watson and Pierce (1998) found that the presence of vegetation that concealed nests dramatically affected disturbance response. Removal of screening vegetation could expose nestlings and increase noise and visual disturbance of adults and juveniles.

7.1.2.2 Effects on Wintering Eagles

Disturbance

Wintering bald eagles use all of the Seattle action areas. Disturbances that cause wintering eagles to flush can result in reduced food intake, increased energy expenditure during critical winter periods, and forced use of marginal habitat (Stalmaster and Kaiser 1997).

Habitat

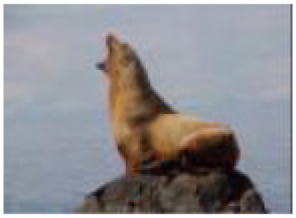
Bald eagles commonly use all Seattle action areas for foraging and nest in all areas except Elliott Bay and North Seattle/Puget Sound action areas. Nesting bald eagles exhibit consistent daily foraging patterns and use of the same perches as they do during the winter (Stalmaster 1987). Perch trees provide bald eagles with some security (Stalmaster and Kaiser 1998). Bald eagles most often forage close to shoreline perch trees (Buehler 2000).

The removal of perch trees from within 250 feet (76 m) of foraging habitat would reduce security and disrupt bald eagle foraging patterns during winter. The result would be reduced feeding and increased energy consumption for both adult and juveniles, which could lead to lower body weights and reduced survival (Hansen and Hodges 1985, Stalmaster and Kaiser 1998).

Conservation Measures

Work timing windows will be imposed on construction activities located near bald eagle nests or known roosting or perching habitat. No work will occur between January 1 and August 15 if the project area is within a 0.25 mile (0.4km) of a bald eagle nest. Similarly, the same work window is observed for projects further than 0.25 mile (0.4km), but in sight of a bald eagle nest. Additional work windows and distances will be followed for roosting and perching habitat and pile driving activities (see CM #1 in **Section 4, Conservation Measures**). Habitat protection measures that minimize the number and size of trees that can be removed will also be incorporated into projects that are within 0.5 mile (0.8 km) of a nest or roosting area. These conservation measures will avoid or minimize impacts to bald eagles.

7.1.3 Killer Whales and Steller Sea Lions

7.1.3.1 Effects of Pile Driving

As with Chinook salmon, bull trout, and steelhead, pile driving and its associated sound pressure levels can injure and affect the behavior of killer whales and Steller sea lions. In-water construction activities, specifically pile driving, may result in elevated sound levels that can affect killer whales and Steller sea lions by causing actual injury, which may result in temporary or permanent hearing loss. Conservative sound exposure thresholds used for describing the level at which sound exposure to broad band impulse sounds, such as pile driving, are 180 dB_{rms} for the potential to cause temporary or permanent hearing loss and 160 dB_{rms} for behavioral disruption.

Sound can also disrupt important biological functions. Killer whales use sound underwater for important life functions including, communicating, finding prey, and navigating. The intensity and persistence of certain sounds (both natural and anthropogenic) in the vicinity of the whales has the potential to interfere with these important biological functions. For instance, the constant production of anthropogenic sound in frequencies that overlap those of biological significance to whales has the potential to mask acoustic signals the species rely upon. It is well documented that killer whales use sound for echolocation (hunting, navigating) and when communicating (Dahlheim and Awbrey 1982, Ford 1989, Barrett-Lennard et al 1996, Ford et al 2000). To accomplish these functions, whales use a wide range of frequencies and have well developed hearing across a broad frequency range of from 1 to 120 kHz or more. Their hearing is most sensitive in the range of 18 to 42 kHz, with peak sensitivity at 20 kHz (Szymanski et al 1999).

The potential for disturbing killer whale and Steller sea lion movements and behavior in Elliott Bay and Puget Sound will be greatly reduced by the suspension of in-water pile driving activities when marine mammals are present in the vicinity. The 160 dB_{rms} sound threshold, described as having the potential to cause behavioral disruption, is the outside boundary to suspend in-water pile driving activities.

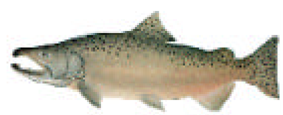
Beyond the boundaries of the 160 dB_{rms} threshold, the attenuated sound levels will likely be audible to killer whales and Steller sea lions, should they be present. Residual sound may be sufficient to cause temporary masking effects out to ambient sound levels but the overall effects to killer whales and Steller sea lions will be minor.

Conservation Measures

An active monitoring program and a protocol to suspend pile driving if marine mammals enter the vicinity is a conservation measure under the Seattle Biological Evaluation (see CM #53 in **Section 4, Conservation Measures**). CM #53 will provide a reasonable degree of certainty that killer whales and Steller sea lions are not exposed to high intensity sound from pile driving at levels that may cause behavioral disruption.

7.2 Effects of the Action on Critical Habitat

7.2.1 Puget Sound Chinook Salmon Critical Habitat



Critical habitat for Puget Sound Chinook salmon within the City of Seattle action areas is limited to the nearshore of Puget Sound, Lake Washington, the Ship Canal, and the Duwamish River. No streams, other than the Duwamish River, are designated as critical habitat. This section describes the effects of the actions (see Table 7-1) on the Primary Constituent Elements (PCEs) present within the action areas. PCEs are physical or biological features that are essential to the conservation of the species. There are 6 Chinook salmon critical habitat PCEs. See **Section 5, Status of the Species**, for a description of each PCE:

- Puget Sound Chinook Salmon PCE #1: Freshwater spawning sites
- Puget Sound Chinook Salmon PCE #2: Freshwater rearing sites
- Puget Sound Chinook Salmon PCE #3: Freshwater migration corridors
- Puget Sound Chinook Salmon PCE #4: Estuarine areas
- Puget Sound Chinook Salmon PCE #5: Nearshore marine areas
- Puget Sound Chinook Salmon PCE #6: Offshore marine areas.

Within each of these PCEs are certain features or elements that are required to support the biological processes for which Chinook salmon use the habitat. Some of these features or elements include water quantity and quality, natural cover, floodplain connectivity, and lack of obstructions.

7.2.1.1 Puget Sound Chinook Salmon PCE #1: Freshwater Spawning Sites

This PCE is not found within the Seattle action areas. Thornton Creek does contain Chinook salmon freshwater spawning sites, but they are not critical habitat.

7.2.1.2 Puget Sound Chinook Salmon PCE #2: Freshwater Rearing Site

Freshwater rearing sites require the following features:

- Water quantity and floodplain connectivity to form and maintain physical habitat conditions that support juvenile growth and mobility
- Water quality and forage supporting juvenile development
- Natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels and undercut banks.

Juvenile Chinook salmon migrating to Puget Sound rear and forage in Lake Washington and the Ship Canal. Most juvenile Chinook salmon use the lake for 1 to 5 months before outmigrating through the Locks. While rearing in Lake Washington, juvenile Chinook salmon are shoreline oriented, using shallow water areas. As juveniles reach a larger size, they disperse to deeper water and begin migration towards the Locks.

Water Quantity

Within the Seattle action areas, designated critical habitat for water quantity relies on upstream influences. Lake levels for Lake Washington and the Ship Canal are controlled by the Locks and are not allowed to fluctuate by more than 2 feet. Inflow to Lake Washington comes from 2 major tributaries, the Sammamish and Cedar rivers. Numerous smaller tributaries also provide water into Lake Washington including Thornton and Taylor creeks. Flows in the Lower Green/Duwamish River are controlled by Howard Hansen Dam.

No proposed projects will remove water from Lake Washington or the Ship Canal. Water quantity will not be reduced by proposed projects. The projects covered under this Seattle Biological Evaluation are tasks that will not be large enough to change the hydrologic regime of Lake Washington, the Ship Canal, or the Duwamish River.

Floodplain Connectivity

No designated critical habitat within the Seattle action areas contains freshwater rearing sites with floodplain connectivity. The Ship Canal is highly urbanized with bulkheads, docks, piers, and other shoreline structures built to protect the commercial infrastructure of the area. The water level in the Ship Canal is controlled by the Locks and fluctuates 2 feet throughout the year. The lowest water level occurs in December and the highest in May. Because of this infrastructure, no floodplain connectivity currently exists. Future project designs may involve increasing shallow water and riparian habitat that could provide some, but minimal, floodplain function, but without huge economic costs, increasing floodplain connectivity would not be feasible.

Water Quality

Water quality within Seattle's designated critical habitat varies with each action area (see *6.1.1, 6.2.1, 6.3.1,* and *6.4.1 in* **Section 6, Environmental Baseline**). Although Lake Washington is highly urbanized, its water quality is very good. This is due to the high quality of water entering the lake as well as the removal of wastewater that entered the lake until the 1960s. Localized water quality problems such as elevated concentrations of metals, bacteria, nutrients, and organic compounds have been found near major stormdrain and combined sewer overflows during storm events.

Water quality in the Ship Canal is generally good due to the high quality of inflowing water from Lake Washington. However, the Ship Canal experiences seasonal temperature and dissolved oxygen problems, as well as occasional problems with fecal coliform bacteria levels. See section *7.2.1.3 Puget Sound Chinook Salmon PCE #3: Freshwater Migration Corridors.*

Construction activities for the proposed projects may result in temporarily decreased water quality. In-water activities, clearing and grubbing, and other bank or shoreline activities will result in short-term increased sediment plumes that may last less than 2 hours. Use of heavy equipment and other construction vehicles poses a risk of petroleum products spilling into the water. Riparian vegetation removal will result in increased sediment input and decreased shade, which can increase water temperatures. Removal of riparian vegetation results in a longer term impact (5 to 10 years) to water temperatures as new vegetation gets established and grows to a size to shade the stream.

Projects that remove creosote-treated timber piles by either full extraction or breaking off the piles at or below the mudline will result in temporary suspension and a long-term increase in creosote-contaminated sediments within the project area.

Forage and Prey Base

Puget Sound Chinook salmon in Lake Washington are opportunistic feeders, consuming a wide variety of prey items and switching quickly to an abundant prey source. In Lake Washington, 2 major prey resources are chironomids and zooplankton. Chironomids are extremely abundant in the nearshore areas of Lake Washington throughout most of the year and zooplankton become abundant in the summer.

Projects along the shoreline of Lake Washington and the Ship Canal that involve the installation, replacement, or maintenance of bulkheads, piers, or hardened shoreline structures will result in simplified shoreline habitat that will reduce forage and prey base species for Puget Sound Chinook salmon. Habitat features such as large woody debris and increased shallow water habitat and riparian vegetation will increase juvenile shallow water rearing habitat.

Natural Cover

Designated critical habitat in the action areas contains very little natural cover. Lake Washington and the Ship Canal are highly urbanized with bulkheads, docks, piers, and other shoreline structures. Large woody debris and other restoration activities to minimize or offset effects associated with hardened shorelines and over-water structures are utilized as much as possible. Within designated critical habitat in Lake Washington, future projects will improve natural cover by placement of large woody debris, removal or set-back of bulkheads, and increasing shallow water habitats.

7.2.1.3 Puget Sound Chinook Salmon PCE #3: Freshwater Migration Corridors

Freshwater migration corridors must be free of obstruction and offer water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival.

Water Quantity, Water Quality and Natural Cover

These are discussed above in section *7.2.1.1. Puget Sound Chinook Salmon PCE #2: Freshwater Rearing Site.*

Obstructions and Barriers

Currently, the only permanent obstruction or barrier to Puget Sound Chinook salmon within the action areas is the Locks within the Ship Canal. The Locks divide the marine and freshwater habitats in the Ship Canal. Passage is possible through the Locks via the fish ladder, large lock, small lock, the saltwater drain, and the smolt passage flumes. Adult salmonids migrating to freshwater primarily pass via the fish ladder and the 2 lock chambers. Juveniles are thought to primarily pass via the smolt passage flumes.

Water temperatures in summer and early fall may be too high and may impede fish migration in Lake Washington and the Ship Canal. Water temperatures along the Ship Canal and in south Lake Union range from 60.8º to 73.4º F (16º to 23º C) between June and September (see Section 6.2.1). In addition, dissolved oxygen regularly drops below 6 mg/L during the summer months when the water temperatures are above (68º F to 70º F [20º C to 21º C]).

Water temperatures in the Duwamish River have increased in the past couple years with temperatures in the summer exceeding 64º F (18º C). High temperatures and low dissolved oxygen can impede juvenile and adult migration through the area.

Docks, both large and small, and other overwater structures are present along the shorelines of Lake Washington and the Ship Canal. These structures may inhibit juvenile salmonids migrating along shallow-water habitats, but have not been found to impede migration. Tabor et al. (1996) found that docks in Lake Washington altered the migration patterns of Puget Sound Chinook salmon, with some juvenile salmon reversing the direction in which they were migrating upon encountering a dock.

None of the proposed actions will result in a permanent obstruction or barrier to Puget Sound Chinook or other salmonids. Construction activities may result in short-term temporary sediment plumes that may impede salmonid migration. However, mitigation measures, like sediment booms or curtains, will be implemented to minimize sedimentation effects. Other construction-related impacts such as clearing and grubbing, may remove some riparian vegetation that could result in decreased shade within the action area. Because only large waterbodies (Lake Washington and the Ship Canal) are designated as critical habitat, the temporary loss of riparian vegetation—until planted vegetation grows to significant size—will not result in increased water temperatures. Pile installation will result in increased sound pressure levels that can impede or prevent salmonid migration. This short-term effect will be minimized through conservation measures such as work timing windows and the use of bubble curtains.

Project designs for projects involving docks and overwater structures will improve existing obstruction and barrier conditions in the long-term. Designs for docks and other overwater structures improve migration corridors for salmonids by minimizing nearshore overwater structure impacts through the use of narrower piers, grating, and the installation of fewer piles. Shoreline restoration and modification projects along the shores of Lake Washington and the Ship Canal will remove bulkheads, retaining walls, and other hard structures and replace them with structures to increase shallow water and habitat complexity that will benefit salmonid migration corridors.

7.2.1.4 Puget Sound Chinook Salmon PCE #4: Estuarine Areas

Estuarine areas must be free of obstruction and excessive predation and offer the following other features:

- Water quality
- Water quantity
- Salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater
- Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels
- Juvenile and adult forage, including aquatic invertebrates and fish, supporting growth and maturation.

The Duwamish River within the City of Seattle Lower Green/Duwamish River Action Area is an all tidally influenced, brackish water environment. This transition zone is very important to outmigrating juvenile Chinook salmon. High densities of juvenile Chinook salmon can be found in this transition zone as juveniles are migrating to Puget Sound.

In the Ship Canal, the estuary has been highly altered due to the construction of the canal and the lowering of Lake Washington and rerouting of the Cedar River system (see **Section 5, Status of the Species**). The Locks structure and its operation influence the physical characteristics of Salmon and Shilshole bays. Juvenile and adult Chinook salmon are forced to move abruptly from one salinity regime to another. Normally juveniles and adults would spend time in the brackish water interface between salinity regimes (acclimation period) before moving into another salinity regime. Because of the Ship Canal, however, little brackish water is available for this.

Obstructions and Barriers

The Duwamish River, like Lake Washington and the Ship Canal, is also highly urbanized with bulkheads, docks, piers, and other shoreline structures. These structures, while not being total obstructions or barriers to migrating Chinook salmon, may impede migration by altering migration patterns by moving juvenile Chinook away from the nearshore into deeper water. Proposed projects under the Seattle Biological Evaluation will improve existing obstruction and barrier conditions in the long-term by increasing shallow water habitat and improving shoreline habitat through modifications of bulkheads, docks, and piers. See section *7.2.1.3 Puget Sound Chinook Salmon PCE #3: Freshwater Migration Corridors*.

Predation

Predators of juvenile Puget Sound Chinook salmon within the Ship Canal action area, upstream of the Locks, include cutthroat trout, bull trout, prickly sculpin, smallmouth bass, largemouth bass, and northern pikeminnow. Below the Locks and in the Duwamish River, cutthroat trout, staghorn sculpin, bull trout, and resident Chinook salmon (blackmouth) are the most prevalent predators. Predation rates have been influenced by the extensive modification of the littoral zone habitats, increase in the population size of predator species, effects of increased water temperature on predator consumption rates, and the introduction of non-native piscivorous fish. Predation of Chinook salmon will be greatest in areas where they aggregate. Within the Ship Canal, juveniles may be most vulnerable to predation as they migrate from Lake Washington to the Locks, pass through the Locks, aggregate below the Locks, and as they rear in the relatively small estuary.

Other predators below the Locks include gulls, harbor seals, and California sea lions. Predation rates of these species on Puget Sound Chinook salmon have been reduced due to changes in operation of the Locks and by removal of nuisance animals and electronic measures to deter predation. The City of Seattle has no control over these measures at the Locks.

Proposed projects for the City of Seattle will help reduce predation in the estuarine environment. While the City does not operate the Locks, future projects in the Ship Canal and the Duwamish River will increase shallow water habitat and habitat complexity important for Puget Sound Chinook salmon survival.

Water Quality and Salinity

Water quality in the Duwamish River has been adversely affected by discharges from public and private storm drains, combined sewer overflows, industrial and municipal wastewater discharges, contaminated groundwater, and spills and leaks that discharge directly to the river from waterfront or overwater activities. Specific water quality concerns included increased water temperatures in the summer and minor decreases in dissolved oxygen. Since 1970, water temperatures have increased about 2º C and have exceeded the salmon migration blockage threshold of 70º F [21º C] during summer.

Salinity is a concern within the Ship Canal. Little brackish water exists around the Locks. Some saltwater is found upstream of the Locks, but is flushed back downstream of the Locks by the saltwater drain. During the summer, a saltwater layer or wedge forms along the bottom of the Ship Canal. This layer combines with summer thermal stratification to make the bottom layers of the water column anoxic. See **Section 6, Environmental Baseline**.

Proposed projects under the Seattle Biological Evaluation will not affect the salinity concentrations within the Ship Canal and Duwamish River. See section *7.2.1.2 Puget Sound Chinook Salmon PCE #2: Freshwater Rearing Site*.

Water Quantity, Natural Cover, and Forage and Prey Base

Water quantity within the Duwamish River estuary is controlled by upstream river systems and future projects will not result in the removal of any water or alter the hydrology of the system. Natural cover with the Duwamish River is limited due to the highly urbanized system. As with Lake Washington and the Ship Canal, future projects will increase natural cover by increasing shallow water and habitat complexity through installation of large woody debris and other habitat features. As in Lake Washington, juvenile Chinook salmon in estuarine areas are opportunistic foragers, feeding on epibenthic and pelagic invertebrates, insects, and small fish. Chinook salmon turn to preying on fish at approximately 6 inches (150 mm) length. Future projects within Puget Sound will not alter the forage or prey base for Chinook. See section *7.2.1.2 Puget Sound Chinook Salmon PCE #2: Freshwater Rearing Site.*

7.2.1.5 Puget Sound Chinook Salmon PCE #5: Nearshore Marine Areas

Nearshore marine areas must be free of obstruction and offer the following features:

- Water quality and quantity conditions
- Forage, including aquatic invertebrates and fish, supporting growth and maturation
- Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels.

Obstructions and Barriers

The Puget Sound nearshore marine area is highly urbanized like the Duwamish River, Lake Washington, and the Ship Canal. Designated critical habitat and proposed project impacts will be similar to that described in section *7.2.1.3 Puget Sound Chinook Salmon PCE #3*.

Water Quality

Water quality in Puget Sound is affected by many factors, including human activities and ocean currents, as well as physical, chemical, and biological processes. The nearshore conditions are affected by human activities such as land-use activities, municipal wastewater discharges, combined sewer overflows, stormdrain discharges, and shoreline erosion. Because many contaminants present in these discharges tend to adsorb to particulate material, the sediment deposited in nearshore areas tends to accumulate contaminants. Areas of concern include the northwest corner of Harbor Island and various locations along the Seattle waterfront.

Future projects covered by this Seattle Biological Evaluation are activities that will not affect water quality within Puget Sound.

Water Quantity

Future projects covered by this Seattle Biological Evaluation are activities that will not affect water quantity within Puget Sound.

Forage and Prey Base and Natural Cover

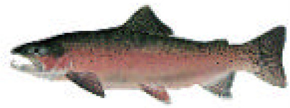
The City of Seattle's future projects will not affect the Puget Sound forage and prey base. Projects will be designed to increase natural cover and shallow water habitat. See section *7.2.1.2 Puget Sound Chinook Salmon PCE #2* above.

7.2.1.6 Puget Sound Chinook Salmon PCE #6: Offshore Marine Areas

Offshore marine areas must have the following features: Water quality conditions offer forage, including aquatic invertebrates and fish, supporting growth and maturation.

No projects will be constructed within the offshore marine designated critical habitat. However, because of the link between nearshore and offshore habitats, there is a potential that future projects may result in a very small change in offshore habitat, but this would be very unlikely.

7.2.2 Coastal-Puget Sound Bull Trout Critical Habitat



Coastal-Puget Sound bull trout designated critical habitat within the City of Seattle action areas includes the Puget Sound nearshore, Lake Washington, the Ship Canal, and the Duwamish River. No streams are designated as critical habitat. All critical habitats in the action areas are considered foraging, migration, and overwintering habitat. There are 8 bull trout critical habitat PCEs (see **Section 5, Status of the Species**, for a complete description of each PCE:

- Coastal-Puget Sound Bull Trout PCE #1: Water temperature
- Coastal-Puget Sound Bull Trout PCE #2: Complex stream channels
- Coastal-Puget Sound Bull Trout PCE #3: Substrate for egg and incubation areas
- Coastal-Puget Sound Bull Trout PCE #4: Natural hydrographs
- Coastal-Puget Sound Bull Trout PCE #5: Groundwater sources
- Coastal-Puget Sound Bull Trout PCE #6: Migratory corridors
- Coastal-Puget Sound Bull Trout PCE #7: Abundant food base
- Coastal-Puget Sound Bull Trout PCE #8: Permanent water.

7.2.2.1 Coastal-Puget Sound Bull Trout PCE #1: Water Temperature

Bull trout have been documented in streams with temperatures from 32 to 72º F (0-22º C) but are found more frequently in temperatures ranging from 36 to 59º F (2-15º C) with adequate thermal refugia available for temperatures at the upper end of this range. Water temperatures within Lake Washington and the Ship Canal during the summer often reach or exceed 72º F (22º C). These temperatures result in a barrier to bull trout entering the Ship Canal. Similar temperatures are found in the Duwamish River, with temperatures exceeding the salmon migration blockage threshold of 69.8º F (21º C). These temperatures in Lake Washington, the Ship Canal, and the Duwamish River limit the use of these waters by bull trout in summer and early fall. Proposed projects covered by this Seattle Biological Evaluation will not result in increased stream temperatures for these waterbodies. Some riparian trees may be removed, but this will not result in increased water temperature.

Maximum water temperatures in Elliott Bay and Puget Sound are about 62º F (16.7º C) offshore and 67º F (19.5º C) along the nearshore. While nearshore temperatures may be too warm for bull trout, prey species, such as Chinook salmon, at this time are not dependent on the nearshore, and, therefore, bull trout will not have to utilize the nearshore. City projects will not affect Elliott Bay or Puget Sound water temperatures.

7.2.2.2 Coastal-Puget Sound Bull Trout PCE #2: Complex Stream Channels

Bull trout require channels with features such as woody debris, side channels, pools, and undercut banks to provide a variety of depths, velocities, and instream structures. Designated critical habitat in the Elliott Bay, Lake Washington, and the Ship Canal contains very little natural cover. Elliott Bay, Lake Washington, and the Ship Canal are highly urbanized with bulkheads, docks, piers, and other shoreline structures. When possible, the City is removing or pulling back bulkheads, and reducing overwater structure impacts within the Ship Canal and in Lake Washington. Large woody debris and other restoration activities are installed or constructed to minimize or offset effects associated with hardened shorelines. Within designated critical habitat in Lake Washington, future projects will increase habitat complexity by placement of large woody debris, removal or set-back of bulkheads, and increasing shallow water habitats.

7.2.2.3 Coastal-Puget Sound Bull Trout PCE #3: Substrate for Egg and Incubation Success

Bull trout do not spawn within any of the Seattle action areas.

7.2.2.4 Coastal-Puget Sound Bull Trout PCE #4: Natural Hydrographs[2]

Within the City's action areas, water quantity for designated critical habitat relies on upstream influences. As noted, lake levels for Lake Washington and the Ship Canal are controlled by the Locks and are not allowed to fluctuate by more than 2 feet. Inflow to Lake Washington comes from 2 major tributaries, the Sammamish and Cedar rivers. Many smaller tributaries also flow into to Lake Washington including Thornton and Taylor creeks. Flows in the Lower Green/Duwamish River are controlled by the Howard Hansen Dam.

No proposed projects will remove water from Lake Washington or the Ship Canal. Water quantity will not be reduced by proposed projects. Increases in impervious surface may increase stormwater runoff, but these projects will not be large enough to change the hydrologic regime of Lake Washington, the Ship Canal, or Duwamish River.

7.2.2.5 Coastal-Puget Sound Bull Trout PCE #5: Groundwater Sources

Springs, seeps, groundwater sources, and subsurface water connectivity are important habitat features for bull trout because they provide cool water refugia. Water

[2] Bull trout require a natural hydrograph with peak, high, low, and base flows within historic ranges, or if regulated, operate under a biological opinion that addresses bull trout. They can also survive in a hydrograph that supports bull trout by minimizing daily fluctuations and departures from the natural cycle of flow levels corresponding with seasonal variation.

temperatures in Lake Washington and the Ship Canal during the summer exceed bull trout temperature thresholds. While bull trout are not expected to be in Lake Washington or the Ship Canal during the summer months, groundwater sources would provide cool water refuge for bull trout. Cool water refugia provide locations that contribute to water quality and quantity.

Proposed projects will not alter any springs, seeps, or other groundwater sources within Lake Washington or the Ship Canal. The Lake Washington and Ship Canal shorelines are highly developed and any proposed projects will improve the aquatic habitat along the shoreline, which could increase groundwater connectivity in these action areas.

7.2.2.6 Coastal-Puget Sound Bull Trout PCE #6: Migratory Corridors

Bull trout need migratory corridors with minimal physical, biological, or water quality impediments between spawning, rearing, overwintering, and foraging habitats, including intermittent or seasonal barriers induced by high water temperatures or low flows.

The only permanent obstruction or barrier to bull trout within the action areas is the Locks within the Ship Canal action area. The Locks divide the marine and freshwater habitats in the Ship Canal. Adult and subadult bull trout migrating to freshwater or the marine waters pass via the fish ladder and the 2 lock chambers.

Water temperatures in summer and early fall are too high and impede bull trout migration in Lake Washington and the Ship Canal. Water temperatures along the Ship Canal and in south Lake Union range from 60.8º to 73.4º F (16º to 23º C) between June and September (see *6.2.1* in **Section 6, Environmental Baseline**). In addition, dissolved oxygen regularly drops below 6 mg/L during the summer months when the water temperatures are above 68º to 70º F [20º C to 21º C]. Water temperatures in the Duwamish River have increased in the past couple years with temperatures in the summer over 64.5º F [18º C]. High temperatures and low dissolved oxygen can impede bull trout migration through the area.

Docks, both large and small, and other overwater structures are present along the shorelines of Lake Washington, the Ship Canal, the Duwamish River, Elliott Bay, and Puget Sound. These structures may inhibit bull trout migrating along shallow-water habitats, but have not been found to impede migration.

None of the proposed actions will result in a permanent obstruction or barrier to bull trout. Construction activities may result in short-term temporary sediment plumes that may impede bull trout migration. However, conservation measures such as work timing windows usually result in construction activities being conducted in summer and early fall when water temperatures are too high for bull trout. In addition, other conservation measures, like sediment booms or curtains, will be implemented to minimize sedimentation effects. Pile installation will result in increased sound pressure levels that can impede or prevent bull trout migration. This short-term effect will be minimized through conservation measures such as work timing windows and the use of bubble curtains.

Project designs for projects involving docks and overwater structures will improve existing obstruction and barrier conditions in the long-term. Designs for docks and other overwater structures improve migration corridors for bull trout by minimizing nearshore overwater structure impacts through the use of narrower piers, grating, and installation of fewer piles. Shoreline restoration and modification projects along the shores of Lake Washington, the Ship Canal, the Duwamish River, Elliott Bay, and Puget Sound will remove bulkheads, retaining walls, and other hard structures, when possible, and replace them with structures to increase shallow water and habitat complexity that will benefit bull trout migration corridors.

7.2.2.7 Coastal-Puget Sound Bull Trout PCE #7: Abundant Food Base

Bull trout require an abundant food base including terrestrial organisms of riparian origin, aquatic macroinvertebrates, and forage fish. Because bull trout are apex predators, as adults and subadults they feed primarily on fish including various trout and salmon species, whitefish, yellow perch, and sculpin. In Elliott Bay and Puget Sound, bull trout also feed on ocean fish, such as surf smelt and sandlance. In freshwater, juvenile bull trout prey on terrestrial and aquatic insects, macrozooplankton, amphipods, mysids, crayfish, and small fish.

Bull trout prey resources are not expected to be appreciably impacted by the proposed projects and activities. Conservation measures will be used for all in-water work to reduce impacts to macroinvertebrates and forage fish from turbidity, sedimentation, and other water quality issues. Fish mix to increase macroinvertebrate production will be installed to cover riprap and fill interstitial spaces. Riparian plantings will increase terrestrial macroinvertebrate input. In the long-term, the bull trout food base should benefit from many City projects.

7.2.2.8 Coastal-Puget Sound Bull Trout PCE #8: Permanent Water

Bull trout need permanent water of sufficient quantity and quality such that normal reproduction, growth and survival are not inhibited.

7.2.2.9 Conservation Measures

All conservation measures incorporated into this document will avoid, minimize, or reduce impacts to critical habitat.

7.2.3 Killer Whale Critical Habitat

Southern Resident Killer Whales critical habitat is limited within the Seattle action areas to Elliott Bay and Puget Sound. There are 3 killer whale critical habitat PCEs.

See **Section 5, Status of the Species**, for a complete description of each PCE:

- Southern Resident Killer Whale PCE #1: Water quality
- Southern Resident Killer Whale PCE #2: Prey species
- Southern Resident Killer Whale PCE #3: Passage conditions.

7.2.3.1 Southern Resident Killer Whale PCE #1: Water Quality

Water quality in Puget Sound is affected by many factors such as human activities and ocean currents. The relatively high water exchange is a key factor in maintaining good water quality conditions in the offshore areas. However, nearshore conditions are affected by human activities such as land-use activities, municipal wastewater discharges, combined sewer overflows, stormdrain discharges, and shoreline erosion. Temperature, dissolved oxygen, and salinity values are fairly consistent throughout the year. Total and dissolved forms of metals are frequently found in Puget Sound waters, but concentrations are generally low.

Construction activities in Elliott Bay or Puget Sound may result in temporarily decreased water quality in the nearshore. In-water activities such as bank or shoreline stabilization or restoration may result in short-term increases in sedimentation. Use of heavy equipment and other construction vehicles pose a risk of petroleum products spilling into the water. However, these activities will impact the nearshore and should not result in water quality impacts to offshore, killer whale critical habitat.

7.2.3.2 Southern Resident Killer Whale PCE #2: Prey Species

Killer whale need prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth. They eat a variety of marine organisms ranging from fish to squid to other marine mammal species. Fish, preferably salmon, are the major food source for Southern Resident killer whales. Chinook salmon comprise approximately 65% of the prey. Other salmonids consumed include pink, coho, chum, sockeye salmon, and steelhead.

City projects in all action areas will result in a variety of impacts to both listed and unlisted salmonids. Construction impacts including stream dewatering, grading, vegetation clearing, etc. may result in increased turbidity, sedimentation, and stream temperatures that may temporarily affect salmonid feeding and rearing. Conservation measures are incorporated into the project to avoid, reduce, and minimize project effects to salmonids. Most projects include habitat restoration or improvement activities—such as increasing large wood, habitat complexity, and removing barriers—that increase or improve spawning and rearing habitat. In the long-term these projects will benefit salmonid populations. Therefore, City projects will, over the long-term, improve or maintain the quantity, quality, and availability of killer whale prey species.

7.2.3.3 Southern Resident Killer Whale PCE #3: Passage Conditions

For killer whales, passage conditions musty allow for migration, resting, and foraging. Most City projects within Elliott Bay and Puget Sound will not result in activities that affect the migration, resting, and foraging of killer whales. A few projects will include pile driving, both impact and vibratory, resulting in increased sound and sound pressure levels that may impede the migratory, resting, and foraging behavior of killer whales. However, conservation measures will be included in these projects that suspend pile driving activities when marine mammals are in the project vicinity. Because of these conservation measures, killer whale migration, resting, and foraging activities will not be affected by City projects covered under this Seattle Biological Evaluation.

7.2.3.4 Conservation Measures

Marine mammal monitoring will occur during all pile driving activities in Elliott Bay and Puget Sound. All pile driving activities will be suspended if marine mammals are seen in the project vicinity and will not resume until all marine mammals have left the area.